Developing Minerals Beneficiation Flowsheets for Eco-efficiency: A Systems Approach

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Thesis presented for the Degree of
MASTER OF SCIENCE IN ENGINEERING
In the Department of Chemical Engineering
Of the UNIVERSITY OF CAPE TOWN

April 2010

Declaration

I know the meaning of plagiarism and declare that all the work in the document, save for that which is properly acknowledged, is my own.

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Abstract

Eco-efficiency has been proposed by the World Business Council for Sustainable Development as a performance indicator framework that contributes to sustainability by assisting corporate decision makers improve the environmental performance of their operations and processes, while also extracting additional economic value. Within the minerals and primary metals industries, a historical legacy of environmental degradation by this sector and the consequent threat of the loss of the social 'license to operate' have resulted in increasingly stringent environmental regulation of metal production processes. The recent vigorous engagement of the minerals industry with sustainability concerns has given credence to this fact. It has also highlighted the need for requisite tools and methodologies for the industry towards addressing these environmental challenges in a systematic and integrated manner.

Given the emergent importance of eco-efficiency within the business community and the urgency with which environmental impacts generated by the minerals industry need to be mitigated, it becomes apparent that there is a need to assess whether eco-efficiency indicators can drive environmental sustainability performance improvement during process design within the minerals industry. This thesis aims to respond to this research need by assessing the strengths and limitations of eco-efficiency indicators as performance metrics in guiding decision making during minerals process design in the interests of environmental sustainability. The ultimate aim of this thesis is to contribute towards improved guidance for process design engineers in the selection of the appropriate tools for more environmentally sustainable design of minerals beneficiation processes.

The systems approach is used as a basis for investigating the strength and limitations of eco-efficiency indicators. First, a more rigorous definition of eco-efficiency indicators specifically for the minerals industry is proposed based on indicators currently offered in the literature. Two case studies are then presented to investigate typical decision situations encountered within minerals process design. These are:

- Process design for the beneficiation of copper metal from a porphyry-type copper sulphide ore (Case study 1), and
- Process design for enhanced water and cyanide recovery from a gold tailings dewatering facility (Case study 2).

Process design alternatives generated on the basis of previous work are used as a basis for eco-efficiency analyses. For each case study, economic and environmental performance assessments are conducted for the design alternatives based on available process data and

an appropriate set of assumptions. These performance assessments are then used to compute eco-efficiency indicators for each design alternative. Eco-efficiency indicators are also compared to more traditional graphical representations of the economic and environmental performance of process design alternatives considered. These comparisons are validated with distinguishability analyses. The sensitivity of these indicators to various process design parameters is also investigated.

The case study investigations confirm that the meaningful application of eco-efficiency indicators is strongly dependent on the decision context in which the process data is generated, i.e. the eco-efficiency indicators to be used for decision making need to be *fit-for-purpose*. It is found that eco-efficiency indicators can be successfully applied in some, but not all minerals process design situations. In particular, it is found that the use of eco-efficiency indicators should be limited to process design cases of positive economic value and positive environmental damage (i.e. cases where an economic *benefit* from the process is expected, at the expense of environmental damage). In other cases, graphical approaches that provide more insights into the environmental-financial trade-offs should be preferred. The analyses also reveal that uncertainty propagation needs to be explicitly considered when eco-efficiency indicators are computed. However, the sensitivity analyses show that eco-efficiency indicators hold significant value in relating economic-environmental performance information to technical process design parameters in a manner that allows for richer communication of performance to explore various design decision trade-offs.

Dedication

This thesis is dedicated to my mother, Ms Sibongile Guma, whose love and support since the very beginning of my academic studies has been a soaring pillar of strength for me.

Mom,

For your sacrifices, patience, endurance, hope and joie de vivre,

Thank you.

Acknowledgements

This thesis would have not been possible without the vision and direction of my supervisors, A-Prof. Harro von Blottnitz and Dr Jennifer Lee Broadhurst. Harro's scholarship, professionalism and support have been an inspiration and a sturdy guidepost for my academic and professional development over the past three years. I also wish to thank Harro for the numerous international travel opportunities he has afforded me – both directly and indirectly; my world has been made richer (and *much* bigger!) as a result. Since the beginning of my postgraduate experience, Jenny has consistently encouraged me push myself to intellectually go beyond mere 'sufficiency for the thesis', and her constructive criticism, sharpness of mind and clarity of thought have truly impacted on my development as a thinker. Special mention also goes to Dr Brett Cohen, who provided me with a firm conceptual foundation at the inception of this research and has been a useful sounding board during the case study analyses. I therefore collectively owe them all a great deal of gratitude and thanks.

Thank you to the E&PSE "Green Group" for creating such a pleasant work environment – not to mention all the "green" tea and lunch socials that have become our hallmark within the UCT Chemical Engineering department! Carol Carr, Mymoena van der Fort and Neli Dili deserve special mention for their efficient administrative assistance. I also wish to extend my thanks to the Minerals-to-Metals consortium for their valuable input into the research, and particularly to Harshad Bhikha, Alex Hesketh and Nomonde Solomon for many a shared "Minerals-to-Metals gauntlet" sessions! Also, the practical contributions of this thesis would not have been possible without the technical input and assistance from various persons and organisations external to UCT: sincere thanks go to Andrew Copeland and Dave Salmon (Anglo Technical Division), Angus Patterson (Patterson & Cooke), Schalk Bekker (Outotec) and Tessa Mackay (Hatch). A special thank you also to Damien Giurco and Gwakisa Mwakyusa for providing the case studies that were the cornerstones of this thesis.

I am extremely grateful to my family for their support and encouragement throughout this experience – to the entire Guma, Ndlovu, Mthethwa and Qobose families, who have been there for me when a respite from this work was sorely needed, I thank you all. And to all my friends – in Cape Town, across South Africa and elsewhere around the globe – for all the laughs in good times and the shoulders to lean on in the difficult ones, my gratitude to you all cannot be overstated. Finally, the Harry Crossley Foundation and the UCT Postgraduate Centre & Funding Office, who have generously provided financial support for the research communicated in this thesis, is gratefully acknowledged.

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List of Acronyms

1,4-DB eq.	1,4-dichlorobenzene equivalents
ADI	aggregated distinguishability index
Ag	silver
Au	gold
CO_2	carbon dioxide
Cu	copper
CIL	carbon-in-leach
DI	distinguishability index
DWAF	Department of Water Affairs and Forestry
EE	eco-efficiency
EIA	Environmental Impact Assessment
ESKOM	Electricity Supply Commission of South Africa
Flash	flash smelting
GPP	Golden Pride Project
HL-SX-EW	heap leaching, solvent extraction and electrowinning
ICME	International Council on Metals and Environment
IIED	International Institute for Environment and Development
LCA	Life Cycle Assessment
MMSD	Mining, Metals and Sustainable Development
NPV	Net Present Value
PSD	particle size distribution
Reverb.	reverbaratory smelting
ROM	run-off-mine
Sb eq.	antimony equivalents
tpa	tonnes per annum
UCT	University of Cape Town
WCED	World Commission on Environment and Development
ZAR	South African Rand

Units of Measurement:

hr	hour (unit of time)
kg	kilograms (unit of mass)
lb	pound (unit of mass)
m	metre (unit of distance)
m^3	cubic metres (unit of volume)
oz	ounce (unit of mass)

ppm	parts per million (unit of concentration)
t	tonne (unit of mass)
wt %	mass or weight percent (unit of concentration)
yr	year (unit of time)

Glossary

Anthropogenic activities: The global man-made production and consumption processes, mostly for the pursuit of economic gain or economic utility.

Antimony equivalents: A life cycle assessment equivalency indicator for the adverse impact associated with the depletion of natural non-renewable resources in the Earth's crust.

Aquatic eco-toxicity: The adverse impact associated with the release of toxic substances (mostly metals) into freshwater sources.

Brownfield operations: Existing operations that are typically subject to retrofits in the process design context.

Decision analysis: The process of elucidating, analysing and evaluating various alternative approaches or options to achieving a defined objective or goal.

Decision context: A set of all criteria describing the *nature* and *consequences* of a decision that needs to be taken to reach a defined objective.

Decision space: A multidimensional (typically two-dimensional) representation of all alternatives available to reach a decision objective, in which the performance of these alternatives can be analysed or evaluated.

Design procedure: A decision analysis procedure whose objective is to produce a final design for a process or product.

Dichlorobenzene equivalents: A life cycle assessment equivalency indicator for the adverse impact associated with the release of toxic substances (mostly metals) into freshwater sources.

Dissipative water consumption: The *net* amount of water consumed by a process (i.e. the difference between the amount of water coming into a process and the amount of water recycled).

Distinguishability analysis: The identification and resolution of situations where uncertainty results in the performance information available being unable to support decision making.

Eco-efficiency: The ability to design products and processes that maximise the economic value derived while minimising the environmental impact associated with that process.

Global warming potential: The extent to which a process can release greenhouse gases that result in an increase in atmospheric temperatures due to radiative forcing.

Greenfield operations: New operations or processes designed to beneficiate a natural non-renewable resource

Minerals beneficiation: The physico-chemical process of mining, concentrating (processing), extracting and purifying minerals or metals from a natural ore resource.

Operational design: Process design activities that are carried out at operational levels of an organisation, often physically limited to a single operation or a single circuit within an operation.

Pareto efficiency: A concept from the field of economics describing the extent to which goods or products within an economy are efficiently allocated.

Resource depletion: The adverse impact associated with the depletion of natural non-renewable resources in the Earth's crust.

Sustainable development: The utilisation of natural resources for human activities in a manner that brings economic benefit without compromising the environment or society in which present and future generations of mankind live in.

Tactical design: Process design activities that are carried out at planning levels of an organisation, whose boundaries may be broader than a single operation.

Nomenclature

A	complete *set* of design alternatives populating the decision space
a	design alternative or process option *i* as an element in the complete set of design alternatives in the design decision space (
B	economic benefit derived from a process option or operation *I* (US$)
C	capital costs (US$)
d	distinguishability index
D	aggregated distinguishability index
E	environmental impact derived from a process option or operation
F	flowrate (mass/time)
G	complete set of performance *criteria* chosen for assessing process options
g	performance *criterion* chosen for assessing process options
i	design alternative or process option
k	type of metal (product) produced by a process option
l	total number of metallic elements contributing to adverse environmental impacts and present within an ore body
M	mass (kg or tonne)
m	total *number* of the types of metal product produced and sold by a process
n	total *number* of design alternatives in the design decision space
O	operating costs (US$)
P	average trading price of a metal product (US$/tonne or US$/oz)
q	*type* of metallic element contributing to adverse environmental impacts and present within an ore body
r	financial discount rate (%)
R	financial revenue (US$)
T	cumulative time period (years)
t	time (years)

Greek symbols:

Ψ	eco-efficiency indicator (B/E, see nomenclature)
ϕ	overall recovery of a metal in a process option (%)
γ	Marshall and Swift cost index ([ ])
ω	concentration of each metallic element in the ore (mass % or ppm)

CHAPTER 1

Introduction

Over the past two decades, the global spotlight on environmental issues has been intensifying. In efforts to address these issues, the concepts of environmental sustainability and sustainable development have emerged and grown in pre-eminence to dominate the global public policy and corporate strategy agendas. The World Business Council on Sustainable Development has proposed *eco-efficiency* as a framework that contributes to environmental sustainability by assisting corporate and technical decision makers identify opportunities for environmental improvement within their operations and processes, while also extracting additional economic value. However, within the minerals and primary metals industries, a knowledge gap exists between the use of eco-efficiency and decision making during the design of mineral beneficiation processes, where significant opportunities for environmental performance improvement exist. This thesis attempts to bridge this gap by exploring the extent to which eco-efficiency indicators can meaningfully guide decision making for the selection of more environmentally sustainable processes during mineral process design. This chapter expands on this premise to establish the overall context for the research carried out in this thesis.

1.1 Background

1.1.1 Environmental concerns and research challenges in minerals beneficiation processes

Growing public concerns over the environmental impacts associated with mining and mineral beneficiation processes have led to a need to improve the environmental performance of these processes. This has been identified as a key global challenge that the mining sector as a whole needs to respond to (IIED, 2001). These challenges are described in this section.

1.1.1.1 Maximising mineral resource efficiency

Rising wealth and exponential population growth has led to an unprecedented rise in the consumption of primary resources globally (Norgate and Rankin, 2002). However, it is also widely acknowledged that these mineral resources are finite, and that the ore reserves from which metals can be economically recovered with currently available technologies have even shorter timespans or ‘years of supply’, as shown in Table 1 (Ayres *et. al.*, 2002; Norgate and Rankin, 2002). Maximising the value returned from mining and beneficiating minerals resources has therefore become a key business case driver in mineral development projects, and has placed an increasing importance in the recovery of other valuable mineral and metal by-products from the process (Scott, 2002).

Table 1: Estimates of a selection of world metal reserves, primary production and years of supply[1]

Metal	Economic ore grade (%w/w)	Reserves (Mt of metal)	Production in 2000[2] (Mt/y)	Years of supply[3,4]	Year of reserve estimation
Iron/steel	30 - 60	65,000	842	77	1995
Aluminium	27 - 29	3,910	24.2	162	1997
Copper	0.5 - 2	320	14.8	22	1997
Lead	5 - 10	65	6.6	10	1997
Zinc	10 - 30	142	8.9	16	1995
Nickel	1.5 - 3	47	1.1	43	1995

While the above is largely an economic challenge, it is important to note that since high-grade ores are processed first in most mining operations, the mined ore grade typically decreases with time as the reserve gets depleted. Figure 1 below shows this trend for copper ore grades in the Unites States. Previous research has shown this decrease in the ore grade to have a dramatic effect on the environmental impact of the mining and beneficiation process (Giurco, 2005; Norgate and Rankin, 2002; Norgate and Rankin, 2000). Compared to high-grade ores, processing of lower ore grades is associated with higher water and energy requirements, greenhouse gas emissions and acidification impacts. Mudd (2007) illustrates this effect using greenhouse gas emissions in the global gold industry (and therefore global warming potential), as shown in Figure 2.

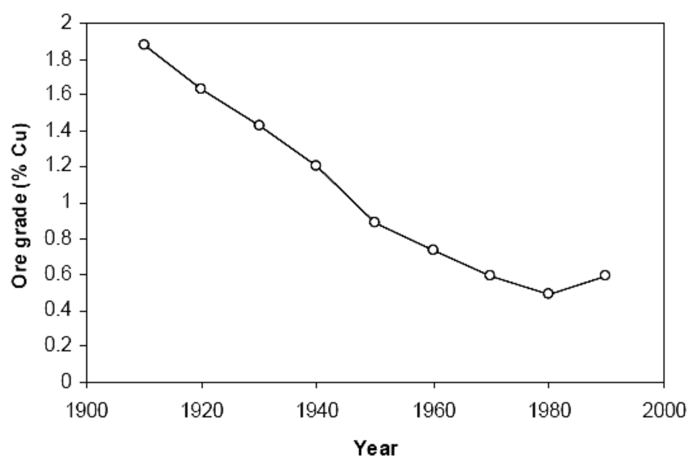

Figure 1: Decline in the average copper grade in the United States[5]

[1] Source: Norgate and Rankin (2002)
[2] Includes primary and secondary metal production.
[3] Assumes consumption rate closely balanced to total production rate.
[4] Assumes no recycling.
[5] Source: Norgate and Rankin (2002)

Figure 2 shows that there exists an inverse relationship between the carbon dioxide emissions and the gold ore grade for the considered operations. This global warming environmental impact, together with others pertinent to the minerals and metals industry, is discussed in greater detail in sections 1.1.1.2 to 1.1.1.4. Such effects place an even greater emphasis on the need to maximise the efficiency of the beneficiation process through developing innovative more efficient mining methods and technologies and creating additional value from by-products during beneficiation.

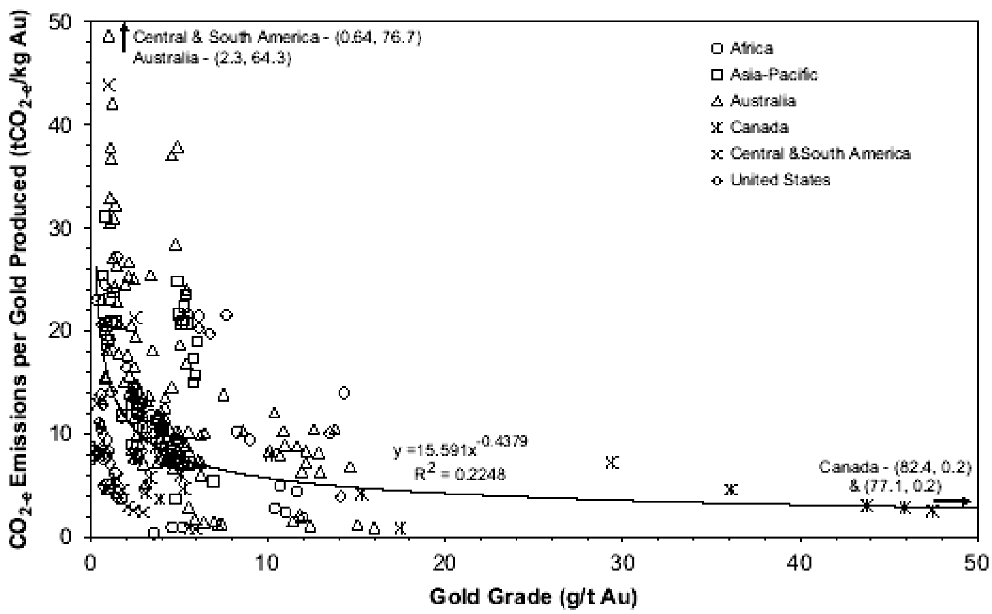

Figure 2: Global variation of CO_2 emissions (equivalent) with gold ore grade[6]

1.1.1.2 Minimising water consumption

The United Nations Food and Agricultural Organisation classifies South Africa as a semi-arid country, receiving on average less than 500 mm of rainfall per annum (Winpenny, 2000), as can also be seen in Figure 3. Water conservation and water management have therefore become of critical importance in the country. Furthermore these concerns, combined with a history of poor water management practices and governance, have become a significant inhibitor of further expansion of the South African minerals and metals industry (DWAF, 2007). The development of water management policies, guidelines and protocols during the design and operation of process plants has therefore become a strategic priority for South African minerals resource firms (Anglo American, 2007).

[6] Source: Mudd (2007)

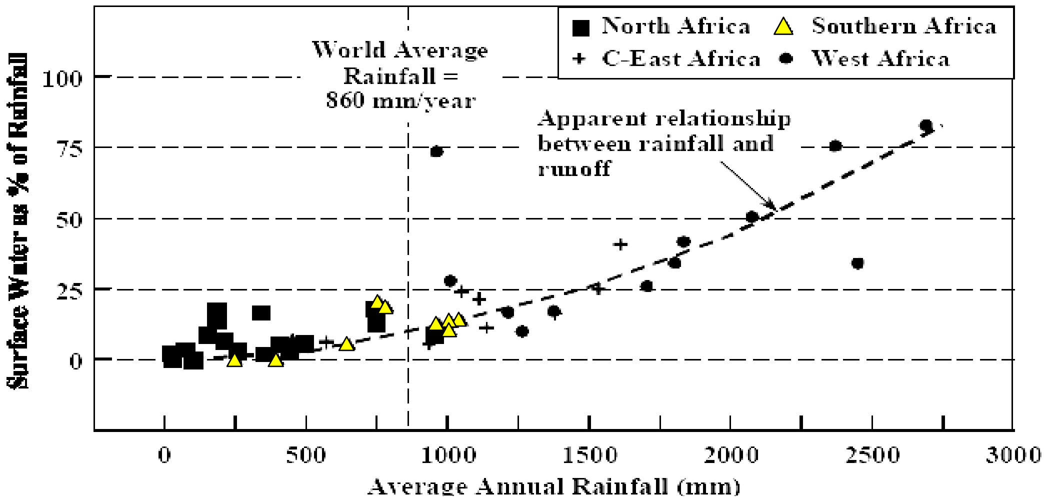

Figure 3: Comparison of Southern African average annual rainfall and surface run-off to other African regions[7]

1.1.1.3 Minimising energy consumption and greenhouse gas emissions

In 2005, the South African mining and minerals industries consumed approximately 17% of all electricity produced in the country (Winkler, 2006). With more than 95% of the total electricity production derived from coal (ESKOM, 2006), this sector is a major contributor to national fossil fuel-derived greenhouse gas emissions and consequent contribution to global warming. To illustrate this, the electricity consumption by the South African mining industry has been compared to mining-related electricity consumption in other regions of the world in Figure 4.

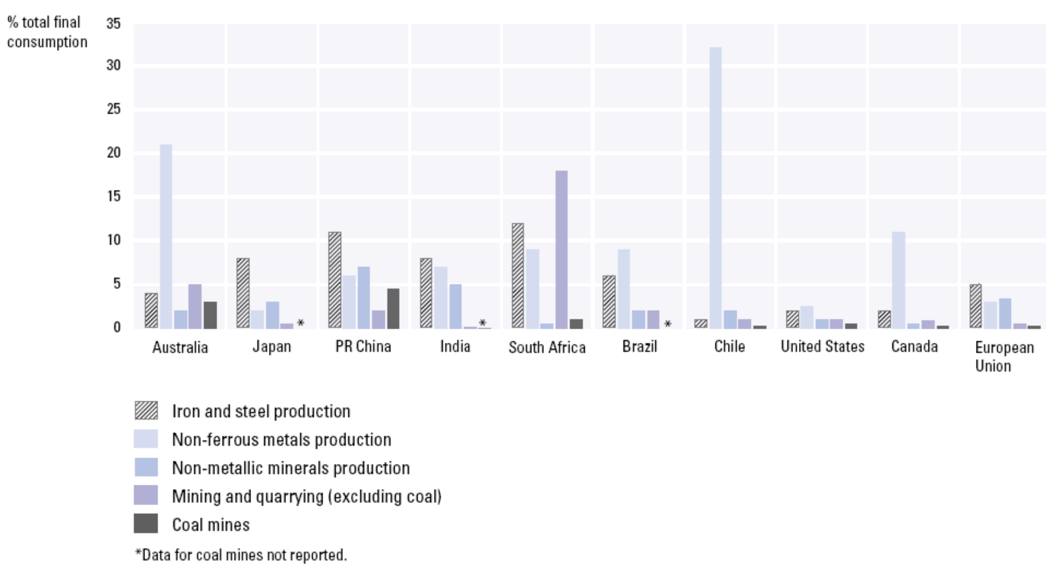

Figure 4: Comparison of South African and global electricity consumption by the mining industry[8]

[7] Source: Ashton *et. al.* (2002)
[8] Source: IIED (2001)

Figure 4 shows that the South African iron and steel, non-ferrous metals, minerals, quarrying and coal sectors account for almost 40% of all electricity consumed in the minerals and primary metals industries[9]. Furthermore, South Africa uses more of its total energy in iron and steel and quarrying industries than any other major mining country in the world (relative to total consumption in the whole sector), and is among the highest consumers of electricity for non-ferrous metals production globally. Underground mines are also significant consumers of diesel and fuel oil, and the combustion of these fossil fuels further increases the sector's contribution to global warming (Giurco, 2005). Recent economic developments in the country, including the emergence of a serious electricity shortage and the possibility of a carbon tax or trading scheme to be imposed on electricity consumption, have added further pressures on the mining and minerals industry to reduce its energy consumption (Sebitosi and Pillay, 2008).

1.1.1.4 Minimising risks emanating from solid wastes

Due to the fact that valuable minerals and metals are often distributed through very large ore bodies and may also be finely disseminated within the host rock, the bulk of the material processed in mining and metallurgical operations contains little value. The minerals and metals industries therefore typically generate large volumes of solid inorganic waste rock (IIED, 2001). Despite the increasing use of land reclamation measures within these industries, the environmental impacts associated with these wastes remains of grave concern. In particular, the generation of acidic or alkaline leachate (leading to further dissolution of toxic inorganic elements and compounds in the waste rock) that results in extensive pollution of groundwater and river systems has been a persistent challenge for the primary metals industries (IIED, 2001). In South Africa, this degradation of water sources is increasingly assuming critical proportions (Hansen, 2004). A simplified representation of leachate generation from mineral waste rock is shown in Figure 5 below.

[9] Crucially, electricity consumed in the precious metals sector (i.e. gold and platinum group metals) – the most energy-intensive of all sub-sectors in the South African minerals industry (Winkler, 2006), has not been included here.

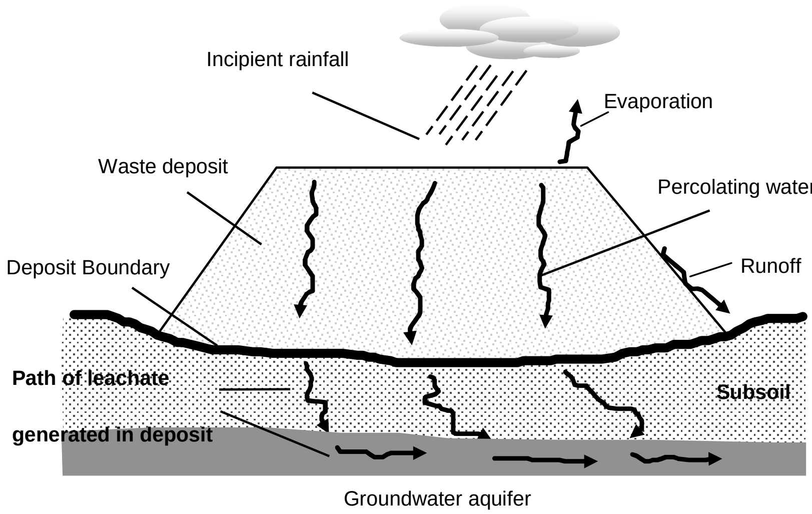

Figure 5: Simplified diagram of leachate generation from mineral solid wastes[10]

1.1.1.5 Integration of environmental considerations into process design

When considered collectively, it can be seen that the above environmental challenges can be reduced to the following pertinent 'themes': the need to *minimise resource consumption* (inputs) and the need to *minimise wastes and emissions* (outputs). While resource consumption and environmental impacts can be reduced in existing metallurgical plants by making modifications to various unit operations in each process (i.e. retro-fitting brownfield operations), it is equally important to consider how new processes (i.e. greenfield operations) can be designed with the above concerns in mind. In this light, the environmental sustainability objective would therefore evolve from improving the environmental performance of *existing operations* to *designing new processes* for optimal environmental performance. This consideration is made even more important given that the process design space often presents the greatest opportunity for improving the technical, economic and/or environmental performance aspects of the process in the entire project life cycle (Stewart *et al*., 2003; Stewart, 1999; Cano-Ruiz and McRae, 1998). As such, the principal area of interest for this thesis lies in the process design-related environmental sustainability considerations within the mining and minerals beneficiation sector.

The above discussion has established that the design of more environmentally sustainable mineral-to-metal processes requires the consideration of multiple and inter-related criteria (relating in particular to valuable minerals, water, energy, wastes and emissions). It can be inferred that this suggests a need for an integrated suite of tools and information sources that will constitute a good basis for environmentally sound and well-informed decision making. Since the engagement of the sector with the issue of sustainable development, the development and successful application of such a suite of tools and information index has

[10] Source: Hansen (2004)

been a persistent challenge (Broadhurst, 2007a; Stewart *et al*., 2003). This arises due to a poor understanding of the links between, *inter alia*:

a) The **information** or data adequately describing material and energy flows in a minerals beneficiation process,

b) **Tools and methodologies** transforming such material and energy balance data to meaningful performance metrics, and

c) The **objectives of decision making** in minerals process design that use such performance metrics to arrive at a certain design decision[11].

The call for the co-consideration of techno-economic and environmental objectives as multiple decision criteria during process design has spawned a plethora of multi-objective tools, systems and methodologies to ensure decision making towards the environmental sustainability of industrial processes. These include Life Cycle Costing (LCC), Life Cycle Assessment (LCA) and Material Intensity Per Service Unit (MIPS), amongst others[12] (Basson and Petrie, 2001; Cano-Ruiz and McRae, 1998). However, Robèrt *et. al*. (2002) maintain that a lack of appreciation for the qualities, differences and linkages between these tools has resulted in much confusion on how best to apply them for effective decision making. This indicates a poor understanding of the underlying *information requirements* required by each tool or methodology and, therefore, the appropriate *context* in which the tool can be applied to enable sound decision making. The latter observation is echoed by Clift (2006) who, in what he calls 'post-normal science', exhorts the engineering discipline to shift from an analytical role to a normative role i.e. where engineers contribute not only to problem-solving (an analytical exercise) but also to framing design problems at an early stage (a normative exercise). Only then, he maintains, can decision making in engineering-based disciplines such as the minerals and metals industries adequately incorporate environmental performance metrics from tools and methodologies such as those mentioned above (which would therefore prompt the generation of the requisite data). This view has been supported by other authors in the process engineering literature (Basson and Petrie, 2007). This need to clarify the links between decision objectives for environmental sustainability and the requisite data using an environmental performance analysis tool therefore forms the key point of departure for this thesis and underpins the overall aim of this work.

[11] Source: Minerals to Metals Research Initiative, University of Cape Town
[12] Basson and Petrie also acknowledge that in a wider decision-making context, engineers need to extend their consideration of multiple objectives during process design beyond the techno-economic and environmental domains to include the social and political milieu, thus adding further layers of complexity to the decision-making process.

1.1.2 A possible solution: The systems approach

In efforts towards addressing the environmental concerns and research challenges of interest in this thesis as outlined in section 1.1.1, the adoption of bolder and more innovative approaches for the development and design of environmentally benign products and processes is now needed, as suggested by Grossmann (2004). Towards this, there is an increasing acknowledgement of the role that process systems engineering can play in assisting chemical and process engineers better understand sustainability issues, thus shifting their thinking beyond providing individual solutions to problems to offering broader pathways to sustainability (e.g. Batterham, 2006; Clift, 2006; Robèrt *et. al.*, 2002). Furthermore, Basson and Petrie (2007) emphasise that improving decision making processes is at the ultimate core of process systems engineering. These arguments therefore position process systems engineering ideally for taking the lead in integrating environmental decision making practice into current minerals process design procedures; and, as such, merit further exploration. A brief overview of the origins and growth of process systems engineering as a field is thus outlined below.

Grossmann and Westerberg (2000) define process systems engineering within the chemical process industry as concerned with “the understanding and development of systematic procedures for the design and optimal operation of process systems”. This need for the optimal design and operation of chemical processes – from conceptual reaction path synthesis through to commercial-scale production – formed the basis for the emergence of this field (Zhelev, 2007). The growth of process systems engineering has been closely linked to progress in the computer sciences domain and recent advances in computing power (Zhelev, 2007). This performance-driven foundation has allowed process systems engineering to enjoy significant accomplishments as a field over the past thirty years, not only in design and optimisation, but also in process operations, for example in scheduling, fault diagnosis and decision support (Grossmann and Westerberg, 2000). As such, its broader focus is well suited for the expanded boundaries of the social, environmental and environmental aspects of sustainable development (Grossmann, 2004).

Despite the breadth of applicability of process systems engineering to a wide variety of sectors, at a conceptual level, process systems engineering is underpinned by a single concept – the systems approach (Singleton, 1985). The systems approach is based on the use of concepts and models for predicting performance and aiding decision making on an engineered system, with an overarching aim to unravel the underlying mechanisms that govern the behaviour of a system (Grossmann and Westerberg, 2000). Singleton (1985) described a fundamental purpose of the systems approach as “to cope with increasingly complex systems”. In light of the preceding discussion, the elements of complexity of interest in this thesis are those that pertain to the need for the consideration of multiple and often competing objectives that need to be achieved by a minerals beneficiation process. In particular, the need to ensure that maximum economic value is derived from a process with as little environmental damage as possible is a well known challenge facing process design engineers (Cano-Ruiz and McRae, 1998).

Indeed, the systems approach has been credited for the emergence of 'sustainability science and engineering' as a new "meta-discipline", harnessing the application of process systems engineering to the fields of physical sciences, engineering, economics and human behavioural studies to develop a new multidisciplinary approach for addressing sustainability issues (Milhelcic *et. al.*, 2003). These systems perspectives to minerals process design are described in further detail in Chapter 2 of this thesis.

1.1.3 Eco-efficiency: A concept and tool towards systemic environmental performance evaluation in minerals beneficiation?

The concept of eco-efficiency was presented to international business and industry at the 1992 Rio Summit[13]. Subsequently, the term "eco-efficiency" was coined by the World Business Council for Sustainable Development (WBCSD) in their landmark report "Changing Course" (WBSCD, 2000) and has since received widespread acceptance both in industry and government (ICME, 2001). The WBCSD (2000) defined eco-efficiency as

"the delivery of competitively priced goods and services that satisfy human needs and bring quality of life while progressively reducing ecological impacts and resource intensity, through the life cycle, to a level at least in line with the Earth's estimated carrying capacity."

In broad terms, eco-efficiency is therefore an environmental performance assessment philosophy that aims to foster the development of products, processes and policies that achieve economic and ecological benefits to society simultaneously, or "create more value with less impact" (WBCSD, 2000). Eco-efficiency forms part of a new environmental management suite of tools and concepts that have gained popularity within academic and business sectors in recent years, such as life cycle assessment, pollution prevention, cleaner production and industrial ecology, among others. It finds a broad and cross-cutting spectrum of application, in both macro-levels (e.g. assessing global eco-efficiency and the eco-efficiency of countries by government) and micro-levels (e.g. assessing the eco-efficiency of a firm and its operations base) of analysis. Eco-efficiency is generically computed through Equation 1 below as defined by the WBCSD, which relates the economic benefit derived from a product or service to the associated adverse environmental impact:

[13] A.k.a. the 1992 United Nations Conference on Environment and Development (UNCED), Rio de Janeiro, Brazil.

$$\textbf{Eco-efficiency} = \frac{\text{Economic benefit of a good or service}}{\text{Adverse environmental impact}}$$

Equation 1: Generic representation of eco-efficiency in equation form

This representation forms the basis for a *quantified* eco-efficiency[14], with which the environmental performance of a process or product can be evaluated. Equation 1 highlights a key feature of eco-efficiency that differentiates it from other environmental performance analysis tools: the simultaneous consideration for **economic** and **environmental** performance in a single metric. This co-consideration of economic and environmental performance as multiple objectives positions eco-efficiency well as a systems-based performance analysis tool. Much work in the field has since been performed to develop the scientific validity and rigour behind the quantified eco-efficiency analysis, most notably in the 2005 special issue of the Journal of Industrial Ecology on Eco-efficiency (e.g. Huppes and Ishikawa, 2005; Dahlström and Ekins, 2005; Kuosmanen and Kortelainen, 2005). However, van Berkel (2007a) asserts that while there are a few examples of eco-efficiency application within the minerals beneficiation sector, notably in Australia, engagement with this concept within the sector has been limited. As such, its potential contribution to environmental decision making theory within this sector, which itself is still in its nascent phases (Petrie, 2007), is yet to be well understood. In this thesis, it is therefore desirable to assess the extent to which eco-efficiency can guide the design of more environmentally sustainable minerals beneficiation processes, and, by so doing, contribute to the development of environmental decision theory for the minerals beneficiation sector. A review of the recent literature on eco-efficiency is provided in Chapter 2 of this thesis.

1.2 Problem statement

The preceding section has recognised that there exists an opportunity to adopt more systems-based approaches in developing environmental performance analysis tools and metrics (such as eco-efficiency indicators) to generate the requisite data for guiding more environmentally sustainable decision making during process design. However, it is equally important that such tools be aligned to the objectives of the decision making process, i.e. each tool should be “fit for purpose”. The research problem to be investigated in this thesis can therefore be summarised as follows:

[14] The concept of a ‘quantified’ eco-efficiency analysis is expanded upon in Chapter 2 of this thesis.

"Despite the acknowledgement and promotion of eco-efficiency as a progressive environmental performance analysis tool towards environmental sustainability in the recent literature, the use of eco-efficiency indicators as environmental performance metrics to meaningfully guide decision making during process design in the minerals beneficiation sector has yet to be explored."

1.3 Objectives and key questions

Having presented the research context to which this thesis aims to contribute to in section 1.1 and summarised the research problem this thesis needs to respond to in section 1.2 above, the broad aim of this thesis is therefore:

To assess the strengths and limitations of eco-efficiency indicators as performance metrics in guiding environmentally sustainable decision making during minerals process design.

For the above objective to be achieved, the following key questions need to drive the research agenda for this thesis:

a) How can eco-efficiency indicators be used to describe the environmental and economic performance of minerals process design alternatives?
b) How can eco-efficiency indicators assist in the framing of decision objectives for minerals process design?
c) What relationships exist between the eco-efficiency indicators as performance metrics, the underlying requisite data and the decision objectives desired during minerals process design?

1.4 Overall approach and structure

This thesis uses eco-efficiency observations and relationships mapped from case studies to contribute to environmental decision making theory-building for the minerals beneficiation sector. Having introduced the thesis in this chapter, Chapter 2 reviews the pertinent literature to crystallise the status quo in the field and verify the initial premises of this thesis. Chapter 3 then draws on key conclusions from the literature to develop the research hypothesis and direct the research design and methodology. Chapter 4 and Chapter 5 present the case studies that encapsulate the different minerals process design situations investigated in this thesis, together with observations on the economic and environmental performance of the various process design alternatives within each case study. Results from these case studies are then discussed comparatively in Chapter 6, after which key conclusions and recommendations to industry and on further work are made.

The overview of the structure of this thesis is presented in Figure 6, alongside key research elements of the case study approach used.

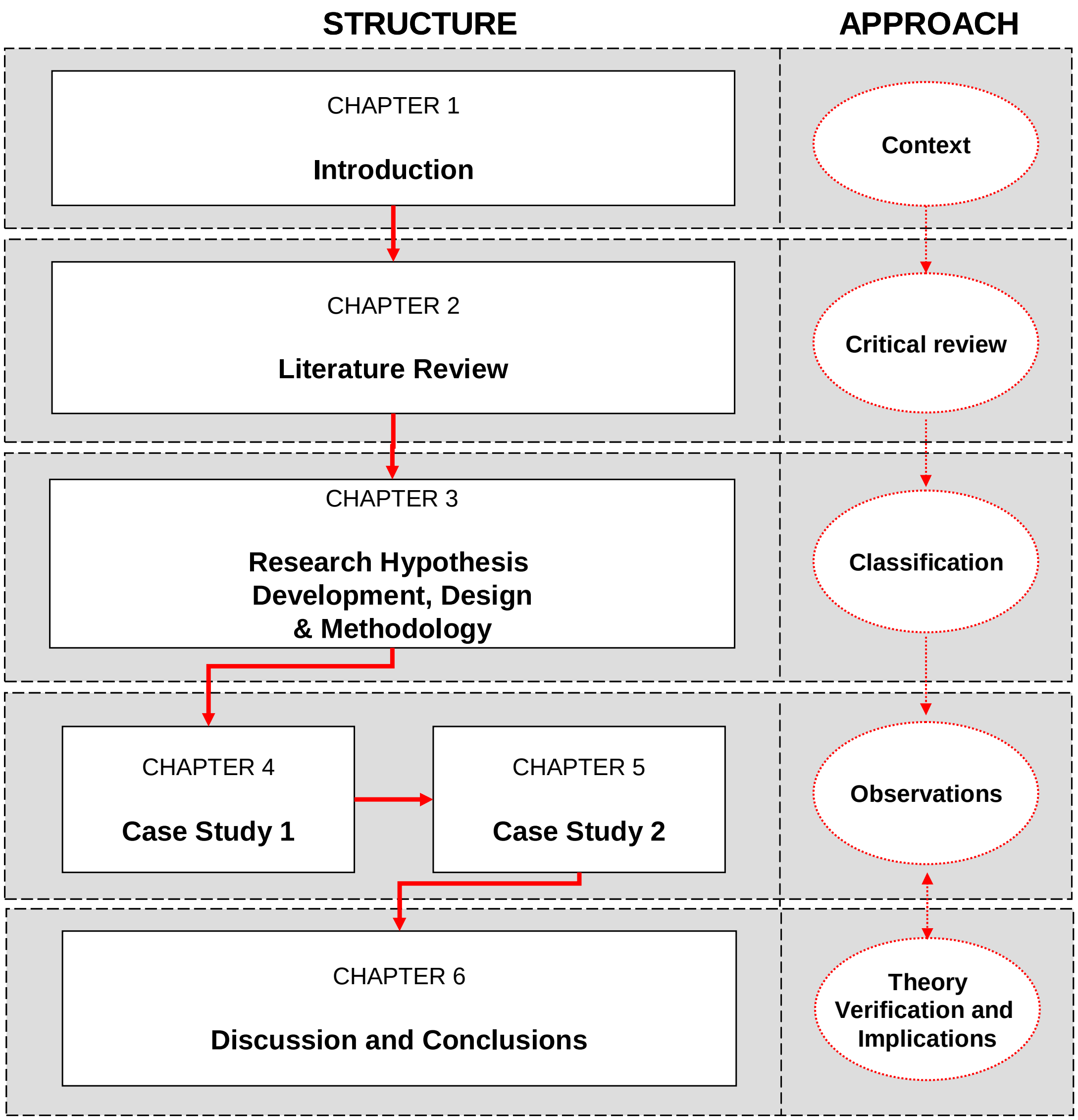

Figure 6: Overall thesis approach and structure

1.5 Scope and limitations

This thesis attempts to map the relationship between eco-efficiency indicators as sustainability performance metrics, their associated information requirements and the decision objectives of minerals process design. While there is scope to apply eco-efficiency indicators across the life cycle of mineral development projects, from exploration phases to post-closure (e.g. van Berkel and Narayanaswamy, 2005), only decisions to be taken during the process design phase are considered in this thesis. Furthermore, the eco-efficiency indicators used in this thesis are selected to reflect the environmental impacts that are of direct concern in the minerals beneficiation sector only.

The approach adopted in this thesis necessitates a careful selection of appropriate case studies which will be representative of all typologies of the theory constructs under consideration (Yin, 1994). While this was pursued as much as possible in this thesis, due to time and data constraints only two case studies were selected and investigated. As such, opportunities for methodological repetition are limited, and empirical or theoretical generalisations can be made only to the extent of the scope of investigation. However, case study research can effectively use inductive reasoning to offer rich and valuable insights for a research study (Yin, 1994), and it is on this premise that the interrogation of eco-efficiency using this approach was sought in this thesis.

1.6 Significance of this thesis

This thesis contributes to the research efforts of the “Minerals-to-Metals” research initiative, a signature research theme established in November 2006 within the University of Cape Town Department of Chemical Engineering. The long-term objective of the research initiative is to improve the understanding of fundamental and systemic research perspectives to the selection, design and optimisation of minerals process and technology options towards achieving environmental sustainability objectives throughout the minerals-to-metals value chain. A systemic component of this aim is to guide the identification, generation and interpretation of appropriate data for decision making in line with environmental sustainability objectives during the design phase in the life cycle of a metallurgical process. This objective therefore defines the key point of departure for this thesis.

CHAPTER 2

Literature Review

2.1 Environmental sustainability in minerals beneficiation

As a response to the global threat of the deterioration of natural resources and its impact on economic and social development, in 1983 the United Nations convened the World Commission on Human Environment and Development (WCED). The commission culminated in the famous report “Our Common Future” (more commonly known as the 1987 Brundtland Report), in which the term “sustainable development” was coined. The following well-known definition of sustainable development was offered:

“*Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs.”*

(WCED, 1987)

The Brundtland definition of sustainable development contains within it two key concepts: the concept of 'needs', in particular the essential needs of the world's poor, and the idea of limitations imposed by the state of technology and social organisation on the environment's ability to meet present and future needs. Through what has been commonly called the “triple bottom line”, sustainable development is underpinned by three forms of sustainability[15] (the now so-called “pillars” of sustainability) as defined by the WCED[16]:

a) **Environmental sustainability**, relating to the sustainability of anthropogenic activities in consuming natural resources and adversely impacting the biosphere,
b) **Economic sustainability**, reflecting the need for human activities to add value to people’s livelihoods, and
c) **Social sustainability**, which encompasses societal expectations.

These three aspects of sustainability have been classically represented using the Venn diagram shown in Figure 7 below:

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[15] While there are some variations in the definition and use of the terms “sustainable development” and “sustainability” in the literature (see Goodland and Daly (1996)) these will be used interchangeably in this thesis.

[16] New models of sustainability are placing increasing importance on a fourth dimension to sustainability, that of *governance*, as to be discussed shortly.

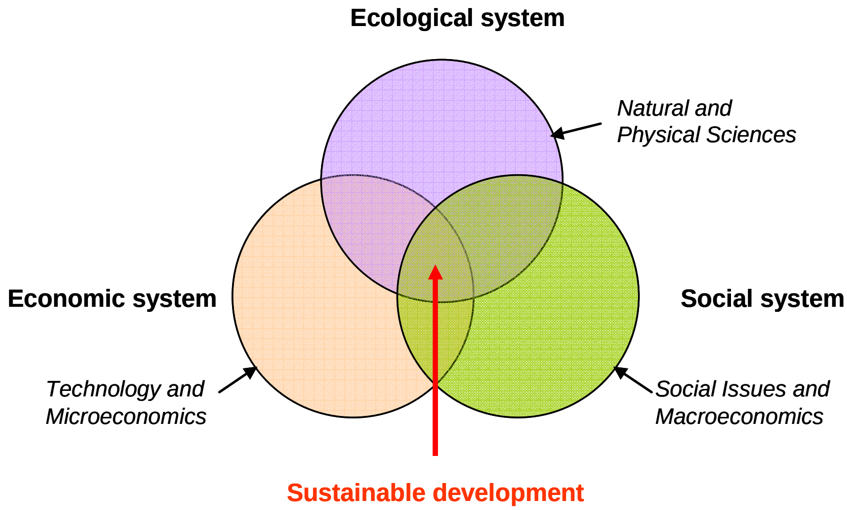

Figure 7: The "Venn diagram" model of sustainability[17]

In contrast to Figure 7, Mebratu (1998) presented the three elements of sustainability (i.e. economic, environmental and social) as 'nested' rather than partially overlapping as shown above – what he calls a dependence of the "cosmos" of sustainability. This view is illustrated in Figure 8 below:

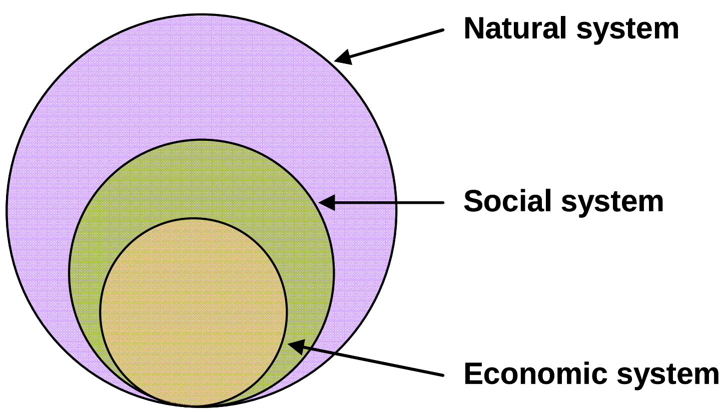

Figure 8: The cosmic interdependence model of sustainability[18]

Other models of sustainability have extended the above models to explicitly consider the three sustainability dimensions as embodying 'capital' or resources that can be harnessed for human activities (Goodland and Daly, 1996). The "five capitals" sustainability model developed through the SIGMA (Sustainability Integrated Guidelines for Management) project in the United Kingdom (Sigma Project, 2007) serves as a good example. The model recognises five key types of capital that underpin sustainability:

[17] Source: Adapted from Baumann and Cowell (1999)
[18] Source: Adapted from Mebratu (1998)

a) *Natural capital* (natural resources that occur in the biosphere as material and energy)
b) *Human capital* (human skills, knowledge and expertise)
c) *Social capital* (institutions such as families, schools and businesses in which human capital is maintained)
d) *Manufactured capital* (material goods and fixed assets such as buildings and equipment)
e) *Financial capital* (wealth created from monetary resources, funds and investments)

The model also adds an extra dimension of *accountability* to these five types of capital, articulating the need to ensure that this capital is used equitably. Indeed, the need for due diligence and transparency in the management of anthropogenic activities to ensure the benefits to society from such activities is an emergent but important theme in the sustainability literature (Mudacumura *et. al.*, 2006; Mebratu, 1998). The model has been shown in Figure 9.

Figure 9: The "Five Capitals" model of sustainability[19]

It is then useful to compare the sustainability models mentioned above. Mebratu (1998) pointed out that while the Venn diagram model has been the most widely used representation of sustainable development, the natural, social and economic systems (or what he calls 'cosmos') can never be analysed independently (where the systems in Figure 7 would be discrete and separate spheres) or even bivalently (with the systems partially overlapping as represented in Figure 7). Rather, they are integrated as shown in Figure 8, with the social and economic systems represented as sub-sets of the natural system. He calls this the 'cosmic misconception' about sustainability, highlighting the failure of the Venn diagram model to

[19] Source: Adapted from Sigma (2007)

capture the interdependence between the economic, environmental and social dimensions to sustainability. These two approaches essentially contrast the normative and positive approaches to sustainability thinking: the Venn diagram model represents the more normative approach, describing a more ideal interaction between the social, environmental and economic facets of sustainability, while Mebratu's approach is concerned much more with the 'reality' of how these sub-systems actually interact. While this view is certainly valid from a theoretical perspective, analytical tools for environmental sustainability performance assessment such as material flow accounting (MFA), life cycle assessment (LCA), eco-efficiency (EE) and ecological footprinting (EF) have so far been relatively successful in relating only the techno-economic and environmental aspects of sustainability (Robèrt, 2000). They therefore lend themselves to more direct application within the overlap between the economic and ecological systems in Figure 7 rather than the interplay in Figure 8 above; realising, however, that the answers they propose are incomplete responses to the sustainable development imperative.

Interestingly, as a relatively new model, the SIGMA sustainability model in Figure 9 seems to represent a hybrid between the Venn diagram and cosmic interdependence models of sustainability. In this model, human, social and manufactured forms of capital are interlinked in a similar fashion to the Venn diagram sustainability model. However, financial capital is seen as completely underpinned by these three types of capital – a distinct characteristic of the cosmic interdependence sustainability model. Furthermore, the human, social, manufactured and financial capital are all nested within the broader sphere of natural capital – the founding premise for environmental sustainability. This model therefore serves to support Mebratu's view of natural capital as the basis for all economic activities and human or social constructs on which society in the broadest sense exist. Indeed, the destruction of natural capital and the consequent rationale for environmental sustainability has been regarded as central to the challenge of sustainable development (Mebratu, 1998; Goodland and Daly, 1996). While the economic dimension of sustainability is a key construct in this thesis (as elaborated on in section 2.3), it is for the above reasons that natural capital and environmental sustainability are of critical concern in this thesis.

Goodland and Daly (1996), in their classical paper entitled "Environmental Sustainability: Universal and Non-negotiable", specifically define environmental sustainability as "the maintenance of natural capital during the pursuit of human anthropogenic activities". Goodland and Daly maintain that the maintenance of renewable and non-renewable natural capital is a key premise for enabling environmental sustainability. This premise is based on three key requirements, namely:

a) The maintenance of renewable natural capital (e.g. air, soil, natural forests, etc.), which implies that wastes and emissions from any anthropogenic activity should be within the maximum limits of the local environment's ability to absorb them without unacceptable environmental degradation,

b) The consumption of renewable natural capital within the regenerative capacity of the natural system that produces them, and

c) The depletion of non-renewable natural capital (e.g. fossil fuel resources) at a rate that is at least equal to the rate at which renewable substitutes are developed and adopted through technological advances and investment.

These environmental sustainability requirements can be traced back to the Rio Earth Summit in 1992, where more than 100 heads of state as well as the international business, activist and academic communities convened to debate and distil global environmental sustainability issues. This conference marked the first concrete milestone where environmental sustainability ceased to be a conceptual preserve of academia and multilateral development institutions, and became a significant point of engagement with global industry and business. Given the need for urgent solutions to the environmental challenges that plague the minerals and metals industries as discussed in Chapter 1, a lively debate has since arisen within the sector as to how to improve the environmental sustainability of mining and minerals beneficiation processes[20]. The most comprehensive and representative address of this need to date has been achieved through the Mining, Minerals and Sustainable Development (MMSD) project, undertaken by the International Institute for Environment and Development (IIED), which culminated in the report *"Breaking New Ground"* that was heralded as the minerals and metals industries' response to the sustainable development challenge (and thus environmental sustainability) at the Johannesburg World Summit on Sustainable Development in 2002. The project aimed primarily to build a global consensus within the sector on the sustainable development issue, assess global mining and mineral use in terms of the sector's transition towards sustainable development and develop an action plan to foster cooperation among all concerned stakeholders for further engagement on the subject. The project was supported by more than 40 commercial and non-commercial organisations in the sector, including nine of the world's largest mining and mineral resource companies. As a key outcome of the initiative, the sustainable development principles that underpinned the overall framework adopted were based on the position that mineral resources must be utilised in a manner that maximises the economic, social, environmental and governance well-being

[20] Petrie (2007) argues that this debate arose due to the fact that the minerals beneficiation sector historically owes its existence to strong global consumer demand for metal and mineral-related products; moreover, this demand is expected to remain in place well into the foreseeable future. At the global level, the challenge was compounded by a supply-demand distortion of the metals value chain along a South-North divide, where the consumer society of rich or developed economies has been (and still remains) the primary driver of demand for mineral and metal products, while developing economies have been the primary suppliers of mineral resources. Therefore, given the deeply-ingrained yet unsustainable consumption patterns of rich-country consumers and the key role the minerals and mining sectors play in the socio-economic development of poor countries, he argues that the call for the maintenance of natural capital by Goodland and Daly (thus implying at least a reduction in the consumption rate of non-renewable ore deposits through either supply-side or demand-side interventions) within the sector was therefore widely regarded as a seemingly unrealistic challenge (IIED, 2001).

of society at large, i.e. natural capital in the form of mineral resources must be used in a manner such that the *overall utility* of society is maximised.

The MMSD project marks a key milestone in the engagement of the minerals beneficiation sector with the challenge of environmental sustainability. While the project "did not try to resolve the many economic, social, environmental and governance issues facing the mining and minerals sector" (IIED, 2001), it did start an important dialogue within the sector towards beginning to develop concerted efforts to solve these above problems. In the meanwhile, however, as stated by Hilson (2001), "there is still plenty of scope for the mining industry to operate more sustainably by adopting strategies that improve environmental protection and promote socio-economic growth". Even the more recent literature (e.g. Petrie, 2007; Azapagic, 2004) still supports such views. This improvement of environmental sustainability as an opportunity for the minerals beneficiation sector therefore forms a key point of departure for this thesis.

The importance of environmental sustainability has been reflected in the emergence of sustainability science and engineering as a so-called new "meta-discipline" in its own right, integrating economic, social and environmental dimensions to the provision of useful goods and services for society in a global context (Milhelcic *et. al.*, 2003). Batterham (2006) asserts that the chemical engineering discipline fulfils a special role within this new meta-discipline, given its capacity to embrace and further develop new disciplines in the pursuit of societal goals. Indeed, Clift (2006) went as far as calling chemical engineers "agents for social change" in their ability to use systems-based environmental analysis and management tools for improving society's quality of life. In Chapter 1 it has been mentioned how process systems engineering, as a chemical engineering sub-discipline, not only constitutes the foundation for current design practice in process industries, but also shows promise in its ability to incorporate such environmental analysis and management tools into process design. This is explored further in the following section, in the context of the minerals and metals industries.

2.2 Process design in minerals beneficiation

2.2.1 Process design: Concepts and theory

Mining and mineral development projects typically go through a series of stages that characterise their *life cycle*, from project conception, through to design, construction, operation, closure and post-closure (Basson and Petrie, 2001). Of interest in this thesis is the design phase of the project life cycle. Turton *et. al.* (1998:2) define process or 'plant' design as "the creative activity whereby we generate ideas and then translate them into equipment and processes for producing new materials or for significantly upgrading the value of existing materials". In minerals beneficiation, this definition can be interpreted as the use of equipment and processes to extract and upgrade the value of mineral-bearing ore deposits into metal and mineral-related products that are useful to society. The various "ideas" Turton and colleagues refer to are expressed in the *design alternatives* that are developed based on technology options that may be used for the production of the desired products. A multidisciplinary design team is therefore tasked with providing a combination of *a priori* knowledge, experience and technical expertise to arrive at the best possible route or "idea" for the production process to be carried out, taking into account all the needs of the involved stakeholders. This task constitutes the design procedure that is the focus of this thesis.

Cano-Ruiz and McRae (1998) define "problem framing" as the first step of the design procedure. At this point, the design problem is defined by means of a concise problem statement, typically constructed after confirming the existence of a mineral ore body that can be beneficiated (Stewart, 1999). Next, possible design alternatives are generated through various design methodologies incorporating both existing design concepts and novel principles or technologies (the "generation" step). Thereafter, the generated alternatives are analysed to characterise and predict their performance against a set of specific design criteria (the "analysis" step). The analysis of process alternatives involves converting the process data generated through material and energy balance calculations to meaningful performance metrics that can be used to evaluate the feasibility of the available process options (Cano-Ruiz and McRae, 1998). This evaluation is then the basis for ranking the performance of the alternatives in terms of their overall attractiveness. Furthermore, opportunities for improving each design alternative are investigated at the sensitivity analyses step. Process design is an iterative procedure, with iterations typically involving generating new alternatives if desired and/or the modifying the framing of the design problem if necessary (Cano-Ruiz and McRae, 1998). The final design is specified once there are no significant improvement opportunities and the design has been characterised to a satisfactory level of detail and accuracy. The design procedure according to Cano-Ruiz and McRae[21] is shown in Figure 10.

[21] There is a wide array of other representations of the design process similar to the above in the literature – in general manufacturing systems (Singleton, 1985), minerals beneficiation (Sudhölter et al.,

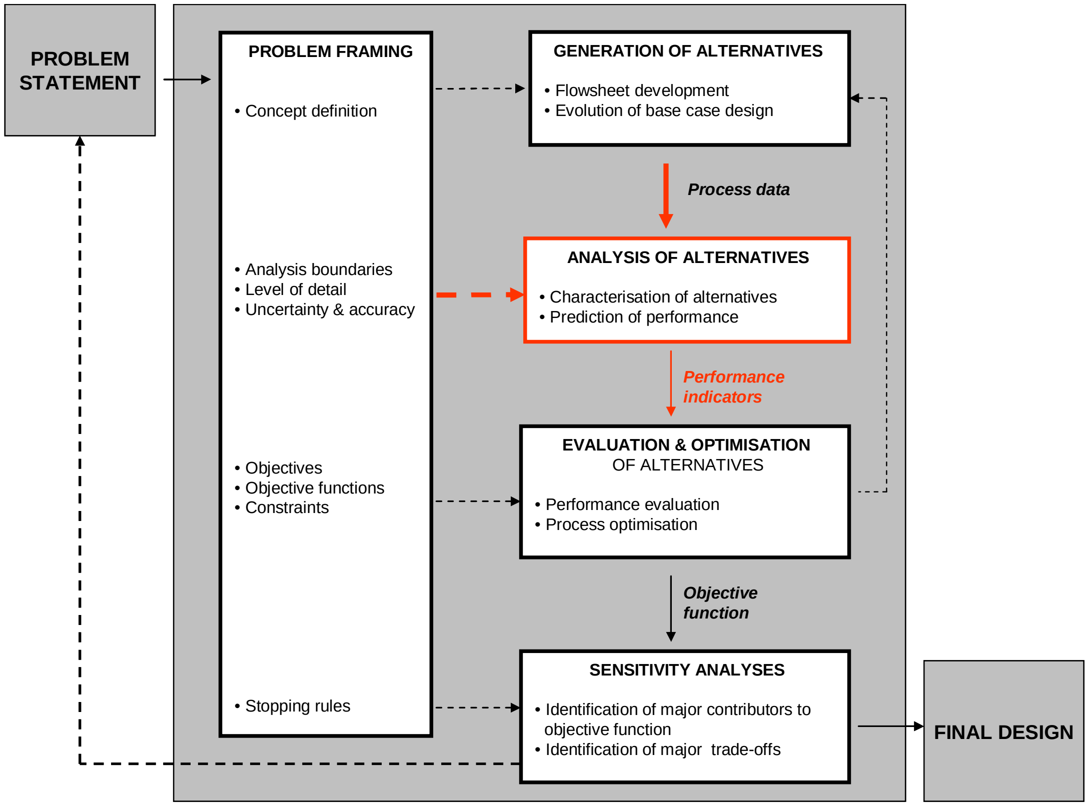

Figure 10: The design procedure as viewed by Cano-Ruiz & McRae (1998)

Since design problems are rarely fully specified at conception, the generation of information at the alternative generation and analysis steps implies that the level of detail and information accuracy changes as the design procedure progresses (Basson and Petrie, 2001). Early phases of process design are usually characterised by a large number of alternatives and relatively broad system boundaries (Stewart *et. al.*, 2003). As such, there is a low level of resolution in the detail of the process alternatives under consideration, and the information available is often associated with a high degree of uncertainty (Basson and Petrie, 2001; Stewart, 1999). During early design phases, design alternatives are typically defined by discrete choices between different technologies or sets of technologies whose operating regimes are represented by typical or average values. As the design procedure progresses, more specific technological choices are made, which increases the resolution of the process detail and reduces uncertainty in performance information (Basson and Petrie, 2001). In tracking this progression these phases are typically classified into the *conceptual*, *feasibility*, *preliminary* and *detailed* phases of process design (Turton *et. al.*, 1998; Douglas, 1988). For the purposes of this study, the design procedure can be divided into conceptual/pre-feasibility, feasibility, preliminary and detailed estimates or studies[22]. For each design phase, the generic design procedure in Figure 10 above is carried out, with the 'final' design(s) at each phase

1996), chemical industries (Chen and Shonnard, 2004) and even corporate or business information processes (Vergidis *et. al.*, 2007).

[22] The "feasibility" phase may sometimes be further divided into a "pre-feasibility" and "feasibility" phase, depending on the industrial sector in which the design procedure is carried out. Further detail on these phases is provided in section 2.2.3.

informing the problem framing and the initial choice of alternatives at the next phase. Process design is therefore a heavily iterative procedure with multiple dimensions of iteration carried out simultaneously (e.g. at the evaluation and sensitivity analysis stages to generate the final design, as well as across design phases to generate accurate and comprehensive information for detailed design producing the final process). These dimensions are shown in Figure 11 below.

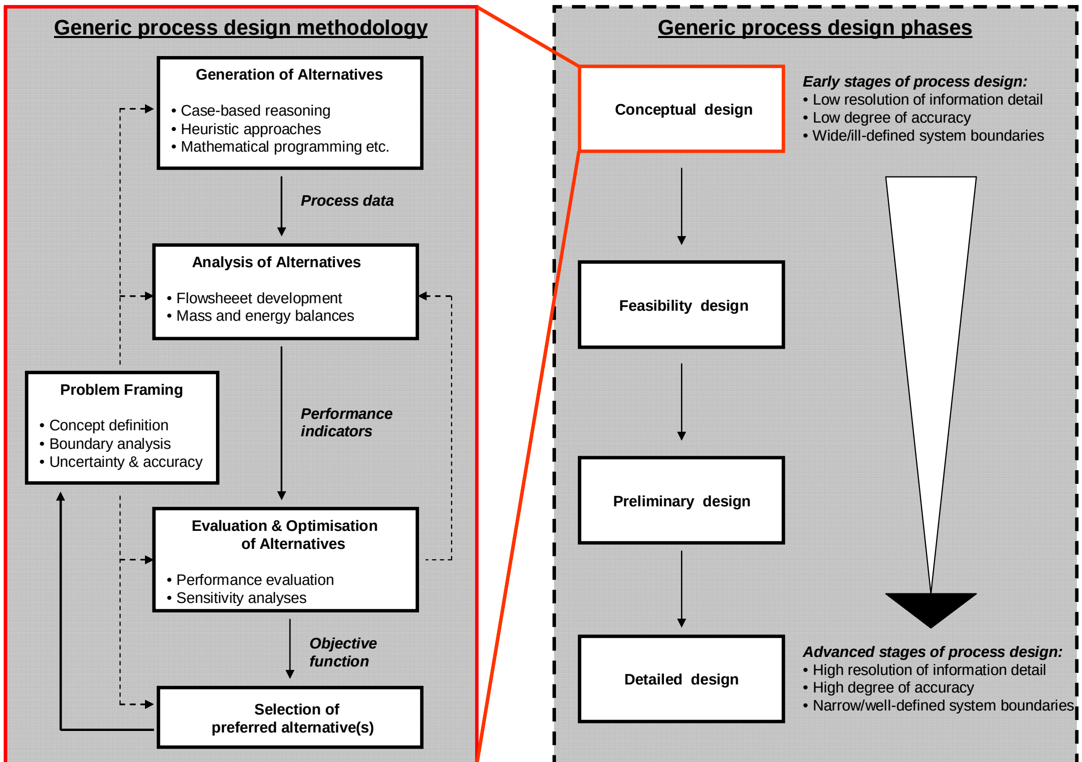

Figure 11: Process design phases and their associated design methodology[23]

Stewart (1999) argues that if the ultimate problem in process design is the need to synthesise a system (in the form of a minerals beneficiation process) that achieves a set of defined objectives (usually in terms of technical, financial, environmental and/or social criteria in the process industries), then process design can be considered as a series of decisions which are taken by the design team of engineers in order to solve the design problem. Due to the environmental sustainability challenge and the consequent need for new ways of thinking, the consideration of process design explicitly in decision making terms has been a subject matter of increasing importance in the process systems engineering literature (Petrie, 2007; Basson and Petrie, 2007; Basson, 2004; Notten, 2002; Stewart, 1999). These contributions have drawn from the field of the management sciences to highlight the need for an understanding of the *context* in which process design decisions are made for environmental considerations to be meaningfully integrated into these procedures. Three generic classifications of such

[23] Source: Modified from Broadhurst *et. al.* (2006)

decision contexts have been offered for commercial enterprises (Wrisberg and Triebswetter, 1999; Weidema, 1998; Rosenhead, 1989)[24]. These are:

a) **The strategic context**, typically entailing planning and capital investment-related decisions,
b) **The tactical context**, containing decisions executed during the design and development of products, technologies and processes, and
c) **The operational context**, in which all decisions relating to operational management, marketing and communication happen.

Process design-related decisions are therefore seen to be seated primarily in the tactical level, with chemical and process engineers regularly making decisions relating to process design and development (Basson and Petrie, 2001). However, it is common practice also to carry out process design activities during the operational phases of the project life cycle, in what is usually termed 'retrofit design', usually for performance improvement (Turton *et. al.*, 1998). While decisions associated with these operational-level design activities could be classified as 'tactical-operational' in nature, in this thesis the terms 'tactical design' and 'operational design' will be used to describe process design activities in the tactical and operational decision contexts. Given its central role in understanding how environmental considerations can be meaningfully incorporated into minerals process design, the concept of the decision context forms a key cornerstone of this thesis.

It has been highlighted above that in making effective and informed decisions for sustainability during process design, multiple design criteria (including environmental criteria) need to be considered. It has also been highlighted that the progression from the tactical to the operational decision contexts across a project life cycle is typically associated with increasing information detail and reduction of uncertainty. The concept of the 'decision space' developed by Basson (2004) is useful in describing this progression. The decision space represents the range of possible outcomes for each criterion that is to be investigated during the design procedure. Basson claims that during the early stages of process design, the alternatives can be defined as discrete choices between different technologies or sets of technologies whose performances are represented by typical or average values. At later or more detailed stages of process design, a potential range of performance values introduced by operating regimes and variations in equipment-related design variables can be determined (what she terms 'discrete feasible regions'). The decision space is now characterised by discrete regions (typically defined by particular technology choices) and continuous regions that represent operating regimes. These two decision spaces communicate the key

[24] It is evident from the literature that this classification of decision contexts can be ambiguous due to variable interpretations of the terms across fields of application (e.g. engineering or management science and operations research). The definitions offered here are those used typically in the systems engineering field. For a more sophisticated classification scheme that circumvents this ambiguity, see the work of Basson (2004).

differences between the 'tactical' (earlier) and 'operational' (later) design decision contexts encountered during process design. These decision spaces are shown in Figure 12(a) and Figure 12(b) for the early and later stages, respectively.

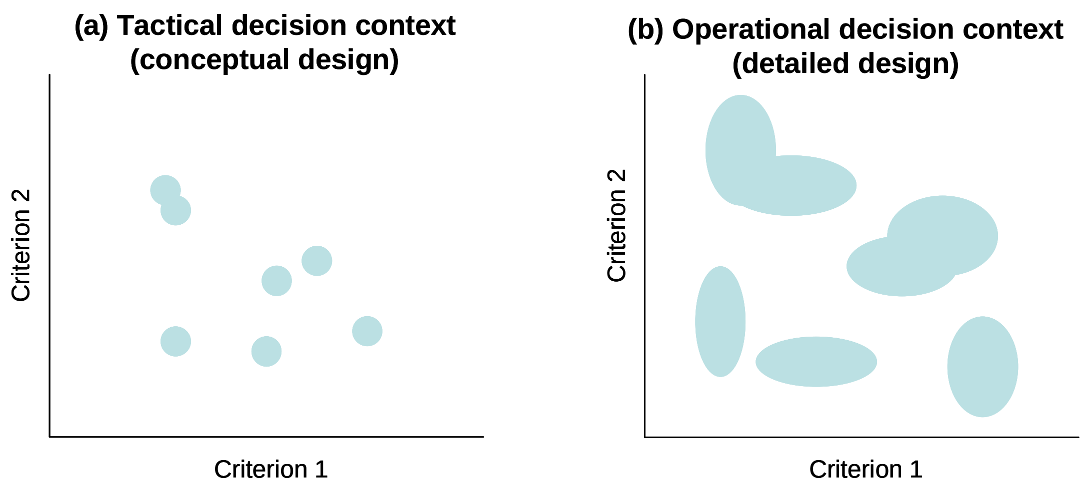

Figure 12: The tactical and operational design decision contexts

Given its central role in understanding how environmental considerations can be meaningfully incorporated into minerals process design, the concept of the decision space as a means for describing the decision context forms a key cornerstone of this thesis. Its conceptual contribution to the research hypothesis to this thesis is further developed in Chapter 3.

Having presented an overview of process design in this section, it is useful to review the status quo in the minerals and primary metals industries towards the design of more environmentally sustainable processes. This is discussed next.

2.2.2 Towards environmentally sustainable minerals process design

The importance of making environmentally conscious decisions on technology choices for flowsheet development at early stages of process design has been recognised in the literature (Notten, 2002; Stewart, 1999; Cano-Ruiz and McRae, 1998). Indeed, Biegler *et. al.* (1997) have observed that early decisions in chemical process design, such as those relating to the choice of reactions and separation processes, "make big differences in our evaluations". Stewart *et. al.* (2003) have echoed this view and extended it beyond technical performance into an environmental paradigm, maintaining that opportunities for major shifts in process performance become increasingly fewer as one progresses through the design phases. Yang and Shi (2000) have also supported this view beyond the design phase to the entire project life cycle, as shown in Figure 13 below. This body of literature therefore makes a solid case for the need to consider design for optimal environmental performance at early phases rather than detailed (later) phases of the design process. This implies developing approaches and frameworks that will provide sufficient information to support decision-making for environmental sustainability during the conceptual phases of process design.

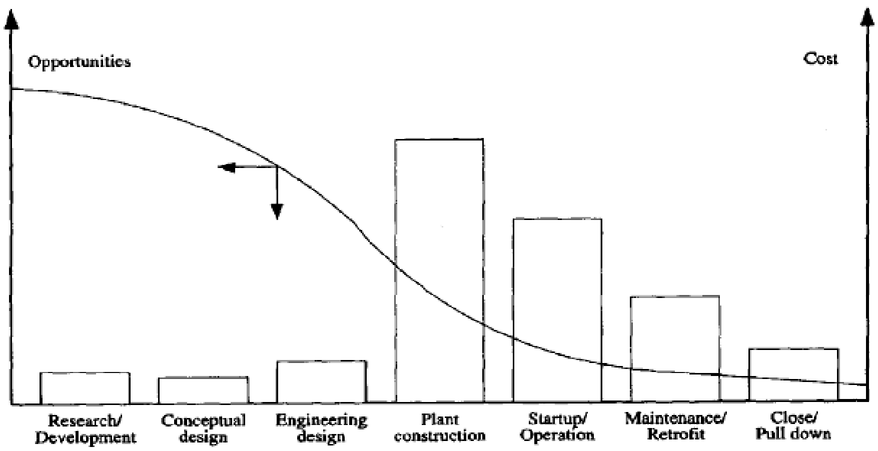

Figure 13: The variation of environmental improvement opportunities and costs across the project life cycle[25]

Various methodologies are available to process design engineers to generate such data sets. The manner in which various process design alternatives are generated is of great importance, given the vast number of existing technologies and processes available to design engineers at the onset of the design procedure (Basson and Petrie, 2001). Furthermore, this step has a direct impact on not only the type of alternatives that are generated, but also on the type of *information* that can be extracted to compute performance metrics – a core concern of this thesis. However, a review of the minerals processing and beneficiation literature reveals little evidence of methodologies for generating design alternatives that facilitate the production of environmental performance information during this step. For instance, while the environmental impact of minerals beneficiation activities is readily acknowledged in the MMSD's Breaking New Ground report, it falls short in providing tactical direction for decision makers on how new or existing process plants can be designed or retrofitted to achieve improved environmental sustainability outcomes. Also, much emphasis is placed on solid waste management (e.g. tailings facilities), with little consideration for other environmental impacts pertinent to the minerals and primary metals industries[26] (e.g. water and greenhouse gas emissions). As a result, most options generation algorithms are still driven by largely techno-economic considerations (e.g. Chakraborty *et. al.*, 2004; While *et. al.*, 2004; Bulatovic and Wyslouzil, 1999). However, there are a few exceptions to this observation; these works are briefly described below.

[25] Source: Yang and Shi (2000)

[26] A later report, "Mining for the Future" by van Zyl *et. al.* (2002), also for the MMSD project, considers water-related impacts more rigorously. However, energy-related impacts such as greenhouse gas emissions are still not considered to any significant detail.

In developing a new method for the generation of process design alternatives *in tandem* with environmental performance information, Stewart *et. al.* (2003) use a heuristics-based approach to incorporate life cycle assessment (LCA) and environmental impact assessment (EIA) into an evolutionary process design framework. This was achieved through augmenting the classical Douglas design hierarchy (shown in Table 2 below) with environmental performance criteria for a zinc refining process based on life cycle assessment indicators. They also incorporated a multiobjective optimisation exercise into this approach to enable trade-offs in performance scores between design alternatives to be explored. The study served as one of the early examples which demonstrated how environmental criteria can be meaningfully related to an established process design heuristics framework within the context of the minerals and metals industries.

Table 2: A heuristics hierarchy for incorporating environmental considerations into flowsheet development in the minerals industry[27]

Design Stage	Step	Douglas Hierarchy	Environmental Design for Minerals
Project Selection	A		Project selection - Comparison of either of possible raw materials, or possible products
Initial Design	0	Input information	
	1	Batch vs continuous	Establishment of reactor-separator trains
	2	Input-output models	LCA of raw materials, products and wastes
	3	Recycle structure	Difference in impacts for concentrated and dilute wastes; identification of waste management philosophies and technologies; linked to EIA
Detailed Design	4	Separation systems	Finalisation of reactor-separator couples and combinations; design of separator systems
	5	Energy integration	Energy mininisation and utilities management

In a later study, this rules-based approach to flowsheet development was applied to model the environmental performance of various major South African and Australian minerals industry sub-sectors on a life-cycle basis (Stewart and Petrie, 2006). An industry-wide comparison of the South African and Australian gold, ferrous metals, non-ferrous metals, platinum group metals, uranium and coal industries was successfully achieved. The generic nature of this approach, coupled with its evolutionary ability to be applied in all stages of the design procedure, holds significant value for its widespread use within the minerals sector in future.

On the evaluation and optimisation of design alternatives, process simulation has emerged as a powerful tool for use by minerals process engineers to aid their understanding of mineral

[27] Source: Stewart *et. al.* (2003)

beneficiation processes (Morrison and Richardson, 2002). As such, it has become a platform for the evaluation and optimisation of minerals process flowsheets (Scott, 2002). Indeed, such has been the impact of process simulation on the technological advancement of design in minerals and metals industries that sensitivity analyses, initially a discrete step in the design procedure (see the overview presented in section 2.2.1), have been fully integrated into the evaluation and optimisation step and are now considered *in tandem* with these steps (Morrison and Richardson, 2002). It is for this reason, therefore, that the evaluation and optimisation of design alternatives (including performing sensitivity analyses) are discussed together in this section.

Herbst *et. al.* (2002) provide a comprehensive overview of mineral processing plant or circuit simulators and differentiate between three types of models that underpin process simulation within this sector: *empirical*, *phenomenological* and *fundamental* models. Steady-state simulators are the most widely used process modelling platforms for design purposes within the mining and minerals beneficiation sector (e.g. JKSimMet®, JKSimFLoat®, SOLIDSIM® and MODSIM®). Circuit or process design is typically based on empirical models, where flowsheet units constitute discrete models or 'black boxes' with defined sets of input and output data. However, none of the empirical simulation packages mentioned above have any significant capacity to characterise the environmental performance of a flowsheet under investigation. The fundamental tasks that these software packages integrate are typically limited only to flowsheet drawing, data analysis, model-fitting, flowsheet simulation and flowsheet performance reporting (Morrison and Richardson, 2002). Databases containing key environmental impact information such as toxicity and acidification potential are currently not available in these simulators. The development of such databases for the minerals sector, as well as effectively integrating these into currently existing simulation packages, is thus a pressing and valuable research need.

The closest examples to achieving this above mentioned integration can be drawn from the chemical process industries. Interestingly, this has been achieved not through the re-development of process design software to include databases and algorithms that support environmentally conscious decision making (which would likely increase research and development costs and stretch computational resources), but rather through coupling the traditional design software tools with environmental performance assessment software and decision support tools, defining various forms of technical information outputs from the design software as inputs for the environmental software package. For example, Cabezas *et. al.* (1999) successfully integrated the ChemCad III® software with the waste reduction (WAR) algorithm to maximise pollution prevention during flowsheet development. Elliott *et. al.* (1996) developed an environmental impact index in MS-Visual Basic™ that could be directly linked to any spreadsheet-based process model. Alexander *et. al.* (2000) used the LCA methodology to characterise the environmental performance of a nitric acid plant, with mass and energy balance information obtained from a HYSIS© model, and using Microsoft Excel™ as the interface between the two models. A similar approach can therefore be envisaged as possible for the minerals and primary metals industries, where the empirical simulators mentioned above can be linked to compatible environmental performance software packages. Such an approach may represent the strongest driver yet for incorporating environmental considerations into standard practice in minerals process design, promoting an integrated approach that harnesses the computational power and tactical advantage of each individual tool or software for *in tandem* use with other tools.

While the above discussion has generally focussed on environmental considerations for the generation and evaluation of process alternatives in minerals process design, of interest in this thesis are the performance indicators computed during the analysis step of the design procedure, as shown in Figure 10 previously. From this figure, it can be seen that the computation of these performance indicators is influenced by two key themes of problem framing, namely the boundary of analysis (which influences the level *of detail*) and the treatment of uncertainty (which relates to the level of *accuracy* associated with the process data). These elements therefore have a direct impact on the *quantity* and *quality* of environmental performance information that can be produced during the analysis of design alternatives from material and energy balances, which in turn exerts significant influence on decision making during process selection. These two design criteria are discussed broadly below.

2.2.3 Key process design criteria for environmental sustainability: A focus on the analysis of design alternatives

The discussions in section 2.2.1 and section 2.2.2 have emphasised the need for integrating environmental considerations into the 'early' or conceptual phases of process design. Within the minerals beneficiation sector, the technical objectives of the design process are typically articulated through a *design basis*. This is a set of technical criteria which the design must satisfy, and forms a formal basis for the final design of the process, equipment and ancillary facilities[28] (Scott, 2002). As the design process progresses from conceptual to detailed design and the alternatives are specified in greater detail, the design basis also becomes more detailed, until there is sufficient information for equipment design and selection to occur (Scott, 2002). It is therefore useful to consider what operational regime defines the context in which the information gathered to describe the design basis is collected. Table 3 below specifies design criteria during the conceptual phase of the design process (Scott, 2002), together with the typical information and data sources for each criterion.

[28] Note that this is different from the economic, environmental (and possibly social) objectives of the design process that have been discussed thus far in this thesis, since the design basis forms a *technical* foundation for what the design must satisfy.

Table 3: Design criteria for conceptual process design in the minerals industry[29]

Design Criterion	Typical information	Typical data sources
General information	Background, context of design work, economic design deliverables	Proposal documents; terms of reference etc.
Metallurgical balance	Preliminary mass balance for all major metals and selected minor metals	Previous design reports for a similar ore body
Operating schedule & throughput	Total annual production time and annual metal product production rate	Average industrial sector values, market analysis documents
Generic process criteria		Previous design reports for a similar ore body, average industrial sector values
Assumptions		Previous knowledge
Recommended testwork	Recommended sampling, bench scale and pilot plant test programs	Key missing metallurgical data

During conceptual design studies, the design basis is based on process data often sourced from similar process plants treating similar types of ore bodies, and preliminary test-work (Marr, 2003). Emphasis at this stage is placed on collecting as much information as possible to determine the metallurgical response of the ore body under consideration (Scott, 2002). Thereafter generic design templates are often set up by metallurgical design houses using the following generalised programme for typical metallurgical flowsheets (Marr, 2003):

1) Ore preparation and dressing,
2) Concentration of value metal minerals,
3) Extraction of the valuable metal(s),
4) Purification of the resultant solutions by removal of bulk impurity elements,
5) Separation and further purification of the principal metal product streams, and
6) Recovery of the valuable metals from solution.

As the design procedure progresses, the decision space evolves from these design templates to the final process that takes into account all aspects that are specific to the particular design problem at hand. The design basis therefore sets an important background to the design procedure during conceptual design, and significantly affects the resolution of detail and data accuracy in the material and energy balance data resulting from flowsheet development and used eventually to compute the performance metrics of the considered design alternatives[30]. These data quality and system resolution effects are described in more detail in section 2.2.3.1 and 2.2.3.2.

[29] Source: Scott (2002)
[30] This is despite the fact that improved knowledge retention from past design projects in the minerals and metals design practice (e.g. the use of case-based design and expert-system based design electronic retrieval methods), coupled with pilot plant technological advances that enable more accurate testwork in the characterisation of ore behaviour through the beneficiation process, are improving the quality of data available to engineers at relatively early stages of process design (Marr, 2003; Scott, 2002).

2.2.3.1 Quality of performance information: Uncertainty considerations

Due to the iterative nature of the design procedure, there is an inherent improvement of the quality of performance information for the various alternatives as the design process progresses. These changes in the accuracy of design information are reflected in a progressive modification of economic design estimates for the proposed process or processes at various phases of the design process. This evolution has been shown in Table 4, with levels of accuracy based on well-accepted 'rules-of-thumb' drawn from largely the chemical process industries.

Table 4: Type and quality of information for the various levels of design[31]

Level of design	Type of information required	Accuracy
Order of magnitude estimate **(Conceptual design)**	Similar previous cost data	±40%
Study estimate **(Pre-feasibility design)**	Knowledge of major items of equipment	± 25%
Preliminary estimate **(Feasibility design)**	Preliminary material and energy balance data; estimates for mechanical, electrical, civil & instrumentation engineering costs	± 12%
Definitive estimate **(Preliminary design)**	Complete material and energy balance data; preliminary engineering drawings	± 6%
Detailed estimate **(Detailed design)**	Complete material and energy balance data; Complete engineering drawings, specifications and site surveys	± 3%

The study of uncertainty as such is still a relatively new niche in the management science literature (e.g. Belton and Stewart, 2002; Functowitz and Ravetz, 1990; Morgan and Henrion, 1990). Given that the accuracy values depicted above result from this section of the literature and are typically associated with economic performance information of the various design alternatives, e.g. Net Present Value (Douglas, 1988), the incorporation of uncertainty considerations into environmental decision making in the resource-based industries is an even more recent addition to the literature (e.g. Basson, 2004; Notten, 2002; Stewart, 1999). Morgan and Henrion (1990) generally distinguish between three different types of uncertainty, as described below.

a) **Uncertainty in empirical parameters**, which pertains to "measurable" properties of real-world systems under study, and as such have "true" rather than "good" or "appropriate" values[32];
b) **Uncertainty in model parameters**, relating to input parameters into a model representation of the real-world system under study (these can be decision variables, value parameters and model domain parameters); and

[31] Source: Modified from Douglas (1988)
[32] Empirical parameter uncertainty can be *aleatory* (i.e. resulting from variability, inherent randomness or unpredictability) or *epistemic* (i.e. owing its existence to statistical variation, subjective judgement, linguistic impression, disagreement and approximation). The key difference between these two types of uncertainty is reducibility - epistemic uncertainty is generally regarded as reducible, while aleatory uncertainty is regarded as irreducible. Refer to Morgan and Henrion (1990) for more details.

c) **Uncertainty in model form**, which accounts for the inherent uncertainty in using a certain type of model (since the use of different types of models can lead to very different results)

Bonano (1995) advocated that the above classification for uncertainties needs to take into account both the substantive and the procedural aspects of decision making. To this end, the author differentiates broadly between *technical* uncertainty and *valuation* uncertainty. Technical uncertainty is concerned with uncertainty in the *consequences* of choosing a certain alternative as a decision analysis outcome, while valuation uncertainty is concerned with assessing the "goodness" of the alternatives considered for the decision at hand. These are shown in Figure 14 below.

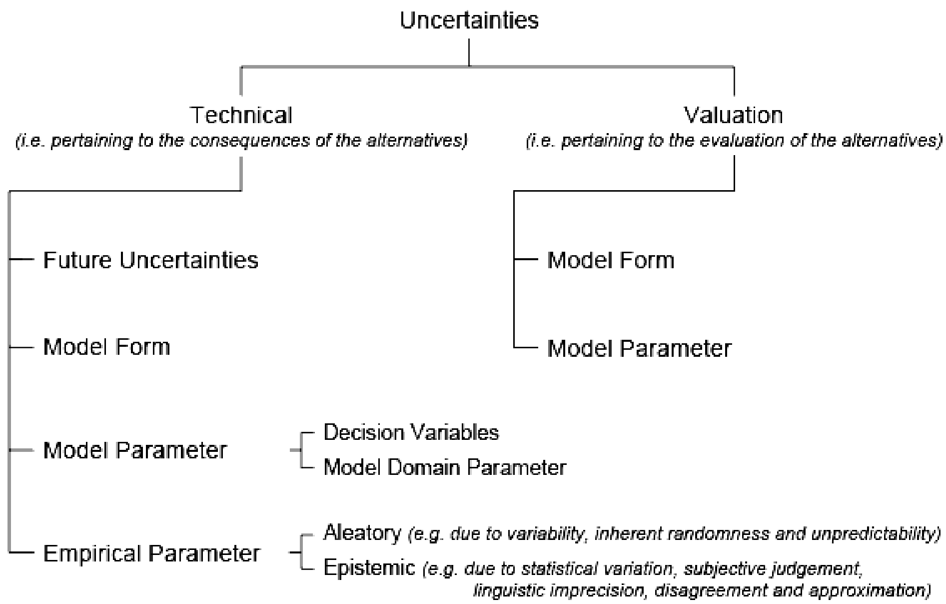

Figure 14: A classification of decision model uncertainties[33]

Each one of these two types of uncertainties requires that the quality of information used allow for a useful elucidation of the performance of the design alternatives considered. To this end, Basson (2004) introduces the concept of *distinguishability*[34], defined as the ability to differentiate between the relative performance values of design alternatives given the uncertainty observed for each alternative's performance information. Basson further refines distinguishability to define an "extent of distinguishability" that differentiates between *complete*

[33] Source: Basson (2004)

[34] Basson (2004) further argues that this concept of "distinguishability" should not be confused with "indifference", since the former refers to the ability of the decision-maker to 'distinguish' between the performance scores of possible alternatives solely as a function of the uncertainty ranges of the performance scores (i.e. excluding any notion of preference), while the latter is a function of the decision-maker's value judgements and by necessity requires the decision-maker's preference information.

distinguishability (when the performance ranges of the alternatives do not overlap), *complete indistinguishability* (when the best estimate of each alternative is contained in the uncertainty interval of another alternative) and *weak distinguishability* (when alternatives are neither completely distinguishable nor completely indistinguishable). When positioned in a process design context, the extent to which the performance of various design alternatives can be distinguishable for a design choice to be made will therefore be dependent on the quality of input information (i.e. design variables) as well as the process model which generates the performance metrics for each alternative. This is a key point of departure for this thesis and will be tested in the case studies, in Chapter 4 and Chapter 5.

2.2.3.2 Quantity of performance information: System boundary and information detail considerations

Mesarovic and Takahara (1975) define general systems theory as "a scientific discipline concerned with explaining various phenomena, regardless of nature, in terms of a formal relationship between the factors involved and ways they are transformed under different conditions". Emphasis is placed on not a "physical object" (e.g. a chemical or social phenomenon), but on a "system" as *a formal relationship between observed features and attributes*. Checkland (1985) argues that while classical scientific thinking or 'the method of science' as a whole has been a powerful tool in creating Western technology and the modern world in the physical sense, its inability to cope with complexity has been a critical shortcoming in the advancement of knowledge of physical, natural and social phenomena. The principal objective of the systems approach is therefore to improve understanding in the physical, social and management sciences by assisting practitioners in these fields of work to deal with complexity within these inter-related fields. Checkland (1985) notes that there is an inherent trade-off in systems thinking that exists between generality and content, i.e. the applicability of systems theory across a variety of disciplines has been seen as coming at the cost of rigour and a low resolution of detail and sophistication in methodology. Within environmental systems analysis, this trade-off typically manifests itself when the *boundaries of analysis* for the system under consideration need to be defined (Notten, 2002). System boundary considerations during process design are thus discussed in greater detail below.

The definition of the system boundary has a significant impact on the computation of performance metrics for a set of design alternatives (Stewart, 1999). The system boundary determines both the economic and environmental bounds of the process or system, i.e. what extent of the product value chain is included in the analysis, and therefore which environmental impacts need to be accounted for in the analysis. While there is little evidence in the minerals and metals literature of the explicit consideration of the system boundary on the performance characteristics of design alternatives, the use of Life Cycle Assessment (LCA) within the sector has proven useful in illustrating the importance of system boundaries[35]. Furthermore, LCA and life-cycle thinking have been successfully and widely applied as a systems approach in the analysis of environmental performance in the process industries (Basson and Petrie, 2007; Clift, 2006; Alexander *et. al.*, 2000; Azapagic, 1999). The discussion below draws from this field of life cycle thinking as a supporting rationale.

Stewart (1999) describes two possible approaches to defining system boundaries for environmental performance analysis in the minerals and metals industry, based on the generic LCA methodology: the *'cradle-to-grave'* and the *'cradle-to-gate'* approach. The 'cradle-to-grave' approach represents the classic LCA approach of incorporating the entire product life cycle into the system boundary, i.e. from resource extraction, through minerals beneficiation, production of finished goods or products to use, recycle and disposal. In this manner, all environmental impacts associated with the production, use and disposal of the manufactured product are captured. The 'cradle-to-gate' approach, on the other hand, includes processes from resource extraction but only up to the point where the extracted materials can be made available to the secondary manufacturing sector (Stewart, 1999), i.e. only resource extraction and minerals beneficiation are included in the system boundary. While the latter approach has the inherent risk of implementing designs that appear to be environmentally superior within this system boundary but merely shift the environmental burden down the product life cycle (Broadhurst *et. al.*, 2006), Stewart (1999) argues that the multiple and pervasive uses of beneficiated minerals and metals products and co-products in the economy would make full cradle-to-grave studies too data-intensive. Furthermore, the use, recycle and disposal of finished metal products typically fall well outside of the minerals beneficiation community's immediate sphere of influence (Stewart *et. al.*, 2003). Therefore, the 'cradle-to-gate' approach to life cycle thinking is considered appropriate for use in this thesis. Figure 15 below illustrates these two approaches.

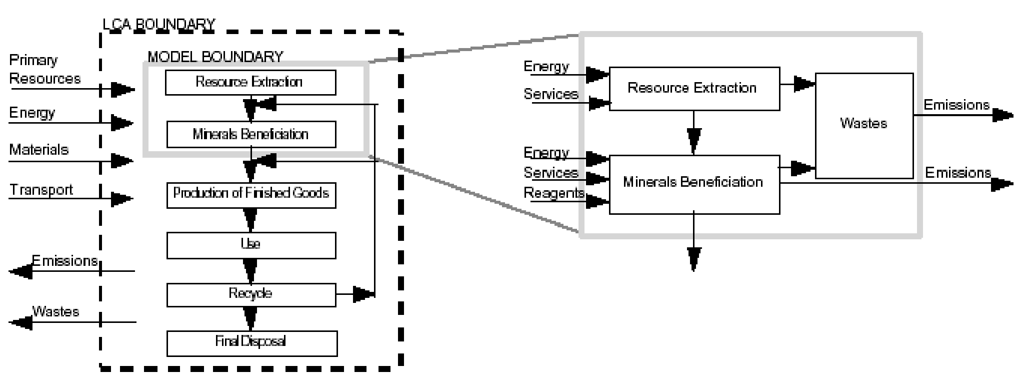

Figure 15: 'Cradle-to-grave' vs. 'Cradle-to-gate' definitions of the system boundary[36]

[35] See the work of Stewart (1999) and Stewart and Petrie (2006) for the application of LCA to the minerals and metals industry; and that of Notten (2002) and Basson (2004) for application to the broader primary resources industries, with a focus on electricity generation.
[36] Source: Stewart (1999)

The model boundary in Figure 15 above represents the 'cradle-to-gate' approach, while the LCA boundary depicts the 'cradle-to-grave' approach. It is important to note that even within the model boundary, resource extraction, minerals beneficiation and waste management are still operationally regarded as separate processes, with little consideration of the systemic effects arising from their interactions during normal operation. Much research and design work in the minerals and metals industry is still performed on circuits rather than processes e.g. comminution circuits (e.g. Huband *et. al.*, 2006; Liu and Spencer, 2004), flotation circuits (e.g. Cisternas *et. al.*, 2004) and wastewater treatment (e.g. Swinkels *et. al.*, 2004). While this has the obvious advantage of focussing the work and driving improvements in fundamental knowledge, it still represents a reductivist approach to process design: there is a lack of understanding of the systemic effects that influence performance *across* unit operations. Broadening these system boundaries from a circuit level to an overall process level may lead to different design outcomes, particularly if economic and environmental performance considerations are taken into account, since these are typically evaluated at a higher systems level (as compared to technical performance criteria, which are only relevant for a particular circuit or even unit operation in the circuit). This argument will be explored in detail in Chapter 4 and Chapter 5.

2.2.3.3 System performance indicators for environmental sustainability

The discussions in section 2.2.3.1 and section 2.2.3.2 have highlighted the importance of understanding the nature of the information available on the environmental performance of design alternatives for decision making during minerals process design. In this section, the literature on suitable environmental indicators is broadly reviewed.

The increasing pressure on firms to be more environmentally accountable has led to immense growth in the scientific development of environmental indicators and maturation into a field in its own right over the past 40 years (Niemi and McDonald, 2004). Much work has been performed recently to characterise environmental indicators (Neimeijer and de Groot, 2008; Kurtz et al., 2001; OECD, 2001). Niemi and McDonald (2004) also provide a comprehensive review of the development and application of ecological indicators. While various environmental indicator frameworks have been proposed in the literature specifically for the minerals and primary metals industries[37] (e.g. Azapagic, 2004; Azapagic, 2003; Azapagic and Perdan, 2000; Hilson and Murck, 2000), there is little evidence of the use of such frameworks within the minerals beneficiation sector to drive environmental performance improvement. Most of these frameworks only facilitate improved corporate environmental *reporting*, a necessary but insufficient step in reducing the environmental footprint of firms. This observation is confirmed by Petrie *et. al.* (2007), who maintain that the only way that sustainability indicators in general can drive sustainability improvements within resource firms

[37] In these works in the literature, these environmental indicator frameworks are often sub-sets of broader sustainability indicator frameworks, which by necessity have to take into account the economic and social aspects of sustainability as well.

is if they inform decision making processes. Towards this, they classify sustainability indicators into two broad categories: *progress* and *transgression* indicators. Progress indicators are defined as measuring progress towards sustainability, while transgression indicators “measure symptoms of environmental degradation or social disenchantment”, i.e. they measure the “current state” of the firm relative to the environment and its social standing. According to Petrie and colleagues, most indicators used within the minerals and metals sector are still transgression indicators. They point out that since decision making is often prospective (e.g. decisions made in the present on future financial targets), progress indicators are more suitable for influencing decision making (and thus drive performance improvement) than transgression indicators. Research towards exploring how the sector can meaningfully use these progress indicators is a key research question that is currently ongoing (Petrie *et. al.*, 2007).

This section has highlighted that the manner in which performance information is generated and communicated during process design has a notable impact on decision making. Given that eco-efficiency indicators are the principal performance metrics which communicate the economic and environmental performance of design alternatives in this thesis, it is now desirable to evaluate the eco-efficiency literature in light of minerals process design. This literature critique is performed next.

2.3 Eco-efficiency: A literature review

2.3.1 Definition and methodological assessment of eco-efficiency

2.3.1.1 Eco-efficiency as a normative sustainability concept

In order to understand the origins of the concept of eco-efficiency, it is necessary to consider another concept which was one of the early anchor points for the business sector in the environmental sustainability debate: *cleaner production*. This term was coined by the Industry and Environment[38] section of the United Nations Environment Programme (UNEP IE) in 1989. It received strong endorsement by Agenda 21, the action programme signed by 150 heads of state and government at the 1992 Rio Earth Summit (UNEP, 1995). Cleaner production (CP) was defined as "the continuous application of an integrated preventive environmental strategy to processes, products, and services to increase overall efficiency, and reduce risks to humans and the environment". According to the UNEP (1995), these practices have been classified into the following themes: *product modification*, *technology modification*, *input substitution*, *recycling and re-use* and *good housekeeping*. Cleaner production was regarded as an *anticipatory* environmental management philosophy, using preventative practices in ensuring the preparedness and responsiveness of firms to mitigating adverse environmental impacts (van Berkel, 2007b). This proactive approach thus represented a higher level of sustainability which superseded 'reactive' approaches to environmental impact minimisation which had dominated sustainability thinking at that time, such as waste minimisation and pollution prevention approaches using so-called 'end-of-pipe' waste treatment methodologies (Basu and van Zyl, 2006). While further developments in this field have since produced other more sophisticated frameworks for sustainability such as industrial ecology, CP is still regarded as a critical milestone that resulted in a real 'step-change' in thinking towards sustainability in this field (van Berkel, 2007b).

However, while the UNEP's central role in international environmental policy making facilitated the widespread acceptance of cleaner production principles by global business after their inception, there still existed a need for quantifiable performance metrics to implement these policies (Robèrt *et. al.*, 2002). Eco-efficiency emerged as one such strategy to achieve this, in translating corporate policy embodying CP principles into meaningful performance metrics that can inform a firm's progress towards sustainability (i.e. cleaner production can thus be regarded as a policy framework in which eco-efficiency goals are implemented or 'operationalised'). The World Business Council for Sustainable Development (WBCSD) has since championed efforts towards improving this translation and packaging eco-efficiency in a manner that is meaningful to businesses and firms.

[38] Presently known as the United Nations Environment Programme Division of Technology, Industry and Environment (UNEP DTIE) based in Paris, France.

At this point, it is useful to recall from Chapter 1 that eco-efficiency was defined by the WBCSD as

"the delivery of competitively priced goods and services that satisfy human needs and bring quality of life while progressively reducing ecological impacts and resource intensity, through the life cycle, to a level at least in line with the Earth's estimated carrying capacity."

Seven key "tactics" were proposed as a basis for achieving eco-efficiency as listed below:

a) reducing the material requirements for goods and services,
b) reducing the energy intensity of goods and services,
c) reducing toxic dispersion,
d) enhancing material recyclability,
e) maximizing sustainable use of renewable resources,
f) extending product durability, and
g) increasing the service intensity of goods and services.

In using these seven tactics, three broad objectives are envisaged: **reducing the consumption of resources**, **reducing the impact on nature** and **increasing product or service value** (WBSCD, 2000). When CP themes are compared to the eco-efficiency tactics, their complementary features are apparent. These objectives can be directly related to the environmental challenges facing the minerals industry that were described in Chapter 1 of this thesis..

Various approaches have been used to position eco-efficiency within the grey and the open literature. For example, the Venn diagram presented in Figure 7 is shown again in Figure 16 below, with eco-efficiency superimposed.

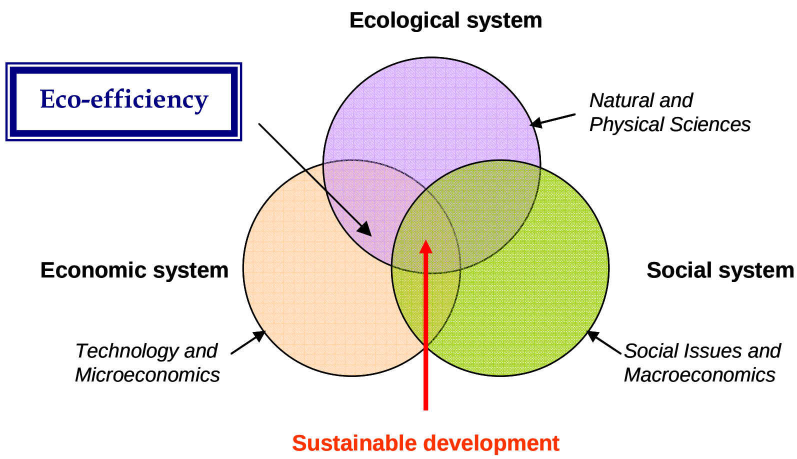

Figure 16: Eco-efficiency within the context of sustainable development using the overlap model[39]

The above representation is useful for explaining the lack of a social sustainability dimension to eco-efficiency. By contrast, attempting to construct an eco-efficiency representation incorporating the economic and environmental spheres using Mebratu's representation in Figure 8 would need to include elements from the social sphere to translate between financial and ecological objectives within the eco-efficiency framework. This might be more correct (but certainly would be more cumbersome), and the notion of eco-efficiency has indeed been criticised for this by some thought leaders within the sustainability community (e.g. Korhonen, 2007; Hukkinen, 2001). This criticism, amongst others, is further expanded upon in the development of the hypothesis for this thesis in Chapter 3.

Recent research efforts have been directed at defining eco-efficiency beyond being only a normative concept to develop consensus and scientific rigour on how eco-efficiency should be quantified (Frigge and Hahn, 2004). This has initiated the field of 'quantified' eco-efficiency analysis, concerned with the development of more standardised eco-efficiency indicator sets or performance metrics (Huppes and Ishikawa, 2005). This field is of central concern in this thesis, and a brief review of this work is provided next.

2.3.1.2 Towards a 'quantified' eco-efficiency analysis

The First International Conference on Quantified Eco-Efficiency, held in Leiden in the Netherlands in 2004, was the first international forum to offer a space for business, academic researchers and the civil service to exclusively debate on what approaches are required to improve quantified eco-efficiency analyses. Significant contributions were highlighted in a special issue of the Journal of Industrial Ecology in 2005[40]. Further dialogue on the subject has since been achieved through a second conference in the series, held in Egmond aan Zee

[39] Adapted from Baumann and Cowell (1999)
[40] Vol. 9, Issue 4, Journal of Industrial Ecology, Cambridge, MA: MIT Press, pp. 1-252

in the Netherlands in 2006[41]. Various approaches to quantified eco-efficiency have been proposed during and since these meetings. The discussion below draws on contributions largely from these two meetings with reference to this thesis.

In defining a quantified eco-efficiency, Huppes and Ishikawa (2005) use the WBCSD eco-efficiency expression to differentiate between four main types of eco-efficiency: *environmental productivity, environmental intensity of production, environmental improvement cost and environmental cost-effectiveness*. Environmental productivity is defined as the production value of a good or service per unit of environmental impact, while environmental improvement cost is defined as the economic cost required per unit of environmental improvement. Environmental intensity and environmental cost-effectiveness are then defined as the mathematical inverses of the environmental productivity and environmental improvement cost, respectively (i.e. numerator and denominator inverted). They then argue that while these are all essentially variants of eco-efficiency, the selection and use of a certain 'type' of eco-efficiency will depend on whether *value creation* or *environmental improvement* is the primary objective sought. The word 'primary' is used since while eco-efficiency by definition seeks to achieve both value creation and environmental improvement simultaneously, various decision situations may dictate which of these objectives is more important. In the process industries, greenfield (new) processes or operations represent good examples of where a value creation or "production" paradigm may be used (e.g. Rüdenauer *et. al.*, 2005), while brownfield (existing) operations may be concerned with mostly improving the environmental performance (e.g. Scholz and Wiek, 2005). This distinction has offered a clarification in eco-efficiency terminology based on decision making, and indicates that various representations of eco-efficiency may be useful depending on the 'primary' decision objective sought.

The above deduction is supported by a rich mix of contexts in which eco-efficiency can be applied. For example, a differentiation can be made between 'micro' and 'macro' level eco-efficiency, where micro eco-efficiency may be concerned with the value created by a firm (or even one product or process within the firm) and the associated environmental impact, while macro eco-efficiency may address the eco-efficiency of a group of firms in an industry (e.g. Dahlström and Ekins, 2005), a geographical region (e.g. Seppälä *et. al.*, 2005) or even a group of industries within an entire economy (e.g. Cha *et. al.*, 2007; Brattebø *et. al.*, 2006). Furthermore, one can differentiate between 'absolute' and 'relative' eco-efficiency, where the latter is an eco-efficiency comparison of a suite of technologies, firms, industries or economies *relative to a specific base case or benchmark* (e.g. Tahara *et. al.*, 2006). Debate on whether multicriteria eco-efficiency (which communicates the eco-efficiency of an entity based on a suite of environmental impacts) or aggregated eco-efficiency (which uses

[41] The Second International Conference on Quantified Eco-Efficiency Analysis, Egmond aan Zee, Netherlands (28 – 30 June 2006). A subsequent gathering has been proposed, the Third Eco-Efficiency Modelling and Evaluation for Sustainability Conference for Guiding Eco-Innovation, occurring in November 2009.

'weighting sets' for economic value and environmental impacts to reduce multicriteria eco-efficiency for an entity to a single performance score) is more appropriate depending on the decision situation is also still ongoing (e.g. Korhonen, 2007; Nieuwlaar *et. al.*, 2005). Such a broad scope of application for quantified eco-efficiency analysis is therefore both a strength and a potential weakness: in such a nascent field, where consensus on the 'correct' application of eco-efficiency has yet to be widely reached, there is a considerable danger in using eco-efficiency inappropriately for decision making. This is a key concern that this thesis aims to investigate, with reference to the minerals industry.

Despite these various approaches and debates, notably, the typology of eco-efficiency indicators by *environmental impact* that have been proposed by the WBCSD are not contested to a significant extent in the literature, indicating some degree of acceptance. The WBCSD broadly classifies the indicators into those that are "generally applicable" (relating to a global environmental concern) and those that are "business-specific" indicators (WBCSD, 2000). Indicators for **energy consumption**, **materials consumption**, **water consumption**, **greenhouse gas emissions** and **ozone-depleting substance emissions** are then recommended as generally applicable indicators all businesses should use, together with a provision for including indicators for **acidification**, other **atmospheric emissions** and **total waste generated** by a business. The recommended economic bases for these indicators are **net sales**, **quantity of goods or services provided**. Müller and Sturm (2001) provided a separate methodology for the development of standardised eco-efficiency indicators, arriving at a similar set of indicators as the WBCSD. These indicators may be viewed as the early developments of a set of heuristics-based indicators which firms or industry sectors can use as a basis for developing more customised indicators. This may be advantageous for the minerals industry, given the numerous data gaps and relatively underdeveloped theoretical frameworks to support environmental decision making that often exist in minerals process design (Chakraborty *et. al.*, 2004; Marr, 2003).

2.3.2 Application of eco-efficiency in minerals process design

As can be expected after the discussion on the origins of eco-efficiency in section 2.3.1.1, much of the early work on engaging with the eco-efficiency concept within the minerals beneficiation sector was an extension of the cleaner production concept. The contributions of Hilson are of particular importance in this regard (Hilson, 2003; Hilson and Nayee, 2002; Hilson, 2001; Hilson, 2000). In his analyses, Hilson consistently maintained that CP needs to be defined for the mining, minerals and primary metals industries in a *specific* and *relevant* manner. His work and that of his colleagues laid a solid platform on which other academic researchers engaged with the concept to eventually make the 'step-change' to interrogating the eco-efficiency concept. The concept, although now relatively well-known within the minerals beneficiation research community, is still actively debated. While some researchers support and promote eco-efficiency as a tool towards sustainability (e.g. Kharel and Charmondusit, 2007; Mäkinen, 2006), others still question its potential for applicability within the minerals beneficiation sector (e.g. Erkko *et. al.*, 2005). This debate is revisited during hypothesis development, in Chapter 3 of this thesis.

The most recent significant attempt at explicitly linking eco-efficiency as used in minerals and metals industries to broader sustainability theory was performed by van Berkel (2007a, 2007b). He provided evidence of the existence of eco-efficiency initiatives within the Australian minerals industry since 1999. These initiatives are then classified into three ‘platforms’ for further development within the sector: eco-efficiency for *existing* operations, eco-efficient design of *future* metallurgical plants and eco-efficiency to foster *innovation in technology routes*. Eco-efficiency was interpreted under five broad themes for the minerals and metals industries: *resource efficiency*, *energy use and consequent greenhouse gas emissions*, *water use and impacts*, *control of minor toxic elements* and *by-product value creation*. In subsequent work (van Berkel, 2007b), the above themes are grouped as *resource productivity themes* and aligned to the cleaner production prevention practices presented in section 2.3.1.1 above. Large-operation industry examples are presented as case studies to illustrate these concepts.

van Berkel pointed out, however, that despite the above successes, there remains a greater challenge in extending the application of eco-efficiency from the largely operational platform to the process design platform within the minerals beneficiation sector, even though some work towards this has been performed (e.g. van Berkel and Narayanaswamy, 2005). Methodologies for translating eco-efficiency indicators to meaningful performance metrics during process design are still considered insufficiently developed. This observation therefore further supports the overall rationale for this thesis, as presented in Chapter 1.

2.4 Concluding remarks

In this chapter, environmental sustainability in the minerals and metals industries has been examined with respect to process design activities in the sector. The extent to which the environmental considerations could be incorporated into minerals beneficiation process design using systems thinking and the systems approach has been reviewed. The literature review has illustrated process design in a decision analysis perspective, highlighting the importance of the need to consider the decision context when selecting performance analysis tools during minerals process design: this has a significant impact on information requirements needed to generate meaningful performance metrics for decision making. The development of eco-efficiency, from a normative concept to a quantified environmental analysis, has also been discussed in the context of minerals process design. Key conclusions from this chapter can be summarised as follows:

Due to the fact that sustainability requires the consideration of often competing economic, environmental and social objectives, evidence from the literature suggests that there is still a wide scope of interpretation of sustainable development, although the minerals and primary metals industries have made progress in at least engaging with this challenge. However, other than a few recent works there is little evidence of the incorporation of environmental considerations into process design and flowsheet development in the literature. This is despite the existence of various environmental indicator frameworks that can guide the industry's practitioners to select appropriate environmental performance indicators with which to assess process design alternatives. This represents a key challenge this thesis aims to address.

Furthermore, it can be concluded from the literature that quantified eco-efficiency shows value as a systems-based environmental performance analysis tool in generating performance metrics that will meaningfully capture and communicate the environmental performance of process design alternatives, despite its relatively limited application to date within the minerals beneficiation sector. However, the decision context and its impact on the quality and quantity of information characterising the performance of these alternatives needs to be explicitly considered. The key question for this thesis and point of departure from this body of literature is therefore whether eco-efficiency indicators can satisfactorily elucidate the environmental performance of the design alternatives investigated across various process design decision contexts. This will be tested in Chapter 4 and Chapter 5 of this thesis.

CHAPTER 3

Research Hypothesis Development, Design and Methodology

3.1 Development and statement of the hypothesis

3.1.1 A case for eco-efficiency in minerals process design

While some of the literature presented in Chapter 2 asserts eco-efficiency as "an important milestone on the sustainable development journey for primary metals production" (van Berkel, 2007b), there is still considerable criticism of the concept regarding in its ability to further sustainable development[42]. This criticism is used as a basis for positioning eco-efficiency for this thesis, and is presented in this section.

The most comprehensive and recent challenge of the eco-efficiency concept has been made by Korhonen (2007). The author seeks to demonstrate that eco-efficiency principles are not suitable for use as sustainability principles, primarily because they lack an overall vision or goal for sustainable development. He maintains that in addition to excluding social sustainability considerations, improving efficiency creates a path dependency and technological lock-in which excludes:

- future solutions, technologies and organisational cultures that support sustainability from development, and
- uptake in the research, management and policy domains.

Improving efficiency is also regarded as potentially going against sustainable development in cases where the increased demand of a more efficient product or service results in a net increase in the environmental impact – the so-called "rebound effect". The author challenges eco-efficiency in its inherent inability to account for an individual's preferences, values and tastes (i.e. eco-efficiency merely *communicates* environmental performance information, but does not assist the decision maker deal with decision trade-offs necessary in *evaluating and selecting* the best-performing product or process). In addition, he points out that these

[42] Also refer to Hukkinen (2001), in his paper entitled *"Eco-efficiency as an abandonment of nature"* for further arguments against eco-efficiency. However, it must be noted that Hukkinen is concerned with the use of eco-efficiency for national and international environmental policy-making, not for firm-level decision making. This application is outside the scope of this thesis and is therefore not considered here.

preferences, values and tastes will change over the long timeframes associated with sustainability analysis, further underlining the impact of this shortcoming.

However, an analysis of the original literature on eco-efficiency shows that the concept was explicitly *not* proposed to present an overall vision for sustainable development (WBCSD, 2000). Indeed, the WBCSD maintains that eco-efficiency is “not an all-inclusive panacea” and is “not a solution to all the problems on the way to sustainability”. Also, given the exclusion of the social aspects of sustainability in eco-efficiency, the concept would have encountered resistance from the onset in acceptance as an overall vision for sustainable development if such a positioning had been sought. The growing use of the concept in environmental and ecological economics, cleaner production, industrial ecology and corporate environmental and social responsibility management (Korhonen, 2007) despite this limitation does not suggest that eco-efficiency is being interpreted as an all-encompassing concept for sustainability, but is possibly viewed as *contributing* to sustainable development. As such, eco-efficiency can therefore be regarded as a *partially useful* sustainability concept – a view that has been echoed by many other environmental theorists (e.g. van Berkel 2007b; Ehrenfeld, 2005; Ekins, 2005). Furthermore, it is becoming increasingly accepted that there is no one approach, tool or methodology that is an all-encompassing goal or strategy towards sustainable development, applicable across all spatial, temporal and institutional contexts – rather, the real value exists in using the strengths of each available sustainability tool or methodology and in the appropriate context for finding sustainability solutions (Petrie, 2007; Hilson, 2003; Robèrt *et. al.*, 2002). For example, multicriteria decision analysis tools have been shown to effectively use environmental performance information to elucidate the values and preferences of decision makers for more sustainable decision outcomes (Basson, 2004). The use of eco-efficiency in conjunction with such tools should therefore strengthen its ability to contribute towards sustainability.

Huppes and Ishikawa (2005) provide a justification for firm-level eco-efficiency (as compared to nationally or globally applicable sustainability concepts) by citing ineffective national environmental public policies brought about by globalisation as the key challenge underpinning poor progress towards international efforts towards sustainable development. Using the Kyoto protocol on carbon emissions as an example, they claim that the accelerated global economic growth and consequent increased international competition for resources of the past few decades has reduced agreeable options for direct environmental policy interventions at the national level despite internationally agreed targets. At current levels of economic growth, national policy trade-offs that have proved sufficient in the past for environmental improvements are now deemed inadequate in maintaining the current environmental quality, let alone improving it. The authors call for the creation of levers of change beyond the nation-state and focussing public policies and private choices where “environmental improvements may be implemented with the lowest economic sacrifice, thus also improving the overall *environmental effectiveness* of measures". This therefore further justifies the implementation of sustainability strategies at the business or firm-level *beyond* the national or global setting. In this light, eco-efficiency is therefore offered as a tool directly empowering firms for proactive environmentally conscious decision-making while providing them with new business-orientated strategic, operational and research opportunities. The well-known eco-efficiency analysis pioneered by the chemical company BASF (Saling *et. al.*, 2002) serves as a good example where concrete evidence was provided that eco-efficiency can be used for not only improving the quality of the firm's decision making process, but also for driving product innovation. There are other examples that have since emerged where innovation as an eco-efficiency benefit has been harnessed (e.g. Mickwitz *et. al.*, 2008; van Berkel, 2007a).

The above critique has therefore proposed that given these persistent national and international public policy challenges, private firms need more direct enabling mechanisms for contributing to sustainable development. This need becomes particularly evident in the process design context for primary metals industries. As mentioned in Chapter 2, the analysis and evaluation of process design alternatives in this sector has traditionally been based on techno-economic criteria without any explicit co-consideration of the environmental profiles of these alternatives. Eco-efficiency can thus be readily used as a conceptual basis for the comparative performance assessment of technologically feasible process alternatives in terms of economic and environmental objectives. Environmentally informed decision-making would therefore be ensured through the simultaneous provision of techno-economic and environmental information to decision makers, together with explicit mention of the implications and consequences of decision maker's choices based on these two sets of criteria.

Eco-efficiency is therefore envisaged to offer several advantages if applied in the above context. Eco-efficiency explicitly considers economic and environmental performance criteria *simultaneously* rather than hierarchically, avoiding multicriteria trade-off analyses and thus simplifying information requirements and the decision space. Also, due to its quantitative basis, it encourages a shift in emphasis beyond merely characterising the environmental profiles of the design alternatives further to identifying possible improvement opportunities even at the early stages of design (Broadhurst, 2007b). Furthermore, it is consistent and compatible with current business principles and practices, and as such it can be readily communicated to business stakeholders and integrated into existing business strategies (WBCSD, 2000). Its organisational origin from the WBSCD (generally regarded as *the* organisation representing the global business community) affords it considerable conceptual legitimacy – an attribute that should not be underestimated in an industry that has historically received much criticism as being resistant to change (Broadhurst, 2007b).

Having made this case for eco-efficiency in furthering economic and environmental performance improvement for the primary metals industries, the first hypothesis for this research can be stated as:

> **"Eco-efficiency indicators can meaningfully communicate the environmental and economic performance of design alternatives in minerals process design."**
>
> ***(Hypothesis 1)***

3.1.2 The use of eco-efficiency in various process design decision contexts

3.1.2.1 Generic representations of eco-efficiency

The environmental and economic performance of design alternatives have traditionally been communicated through both numerical and graphical approaches (Kuosmanen and Kortalainen, 2005). These are described below.

When eco-efficiency performance is represented by a numeric indicator, if the financial revenue derived from a process alternative *i* is R_i and the total annual cost is C_i, then the economic benefit extracted from the process $B_i = R_i – C_i$, so that the eco-efficiency indicator Ψ_i can be defined as in Equation 2 below:

$$\psi_i = \frac{B_i}{E_i}$$

Equation 2: Eco-efficiency defined for mapping onto a 2-dimensional design space

where E_i is the environmental impact under investigation and B_i is defined in monetary units.

The above definition is consistent with the WBCSD definition of eco-efficiency indicators, and will be the basis for all eco-efficiency indicators computed in this thesis.

However, given the need to develop the rigour for this definition of a quantified eco-efficiency (Huppes and Ishikawa, 2005), a meaningful comparison of the 'classical' eco-efficiency metric as defined in Equation 2 with other forms of economic and environmental performance representation is necessary. Graphical approaches to such performance representations have been used extensively in the literature (e.g. Michelsen *et. al.*, 2006; Kuosmanen and Kortalainen, 2005; Saling *et. al.*, 2002). These approaches are historically based on Pareto efficiency[43], a neoclassical economics concept which was developed by welfare theorists

[43] After Vilfredo Pareto, a well-known 20th-century economist who is credited with the development of this concept.

based on an 'efficient' allocation of goods within an economy (Jollands, 2006). The economy is considered efficient with respect to two produced goods when it is impossible to increase the production of one good without a reduction in the production of the other good (Shukla and Deb, 2007). This 'benefit-benefit' paradigm (where the two goods represent an economic benefit to society) has been extended to environmental resource economics, where the environment is explicitly considered as a public good (Jollands, 2006; Kuosmanen and Kortalainen, 2005). Clift (2006) shows how the concept can be applied within the chemical engineering process design community, in the context of a private firm concerned with an inverse 'cost-cost' paradigm with the objective of minimising the economic cost and environmental impact associated with a process. The two axes form the decision space on which the performance of the various alternatives can be depicted, as described in Chapter 2. On each axis, either absolute or relative values can be plotted. These are briefly described below.

The environmental and economic performances of process design alternatives represent the two decision criteria of interest in this thesis, with which the decision space can be defined. The 'best-performing' alternatives in terms of these criteria can be described as *Pareto-optimal* (i.e. where further improvement of environmental performance can only be achieved at the expense of economic performance, and vice versa). The locus of Pareto-optimal 'solutions' forms the *Pareto frontier* or *Pareto set* and thus represents the most superior range of solutions in the defined domain of each criterion (Clift, 2006; Mattson and Messac, 2003). Such a representation is shown in Figure 17 below for a range of design alternatives yielding an economic benefit B and environmental damage E, framed by a two-dimensional decision space such that $B_1 < B_i < B_2$ and $E_1 < E_i < E_2$ for any design alternative *i*. The numerical eco-efficiency indicator defined in Equation 2 can be inferred from the decision space as the slope of the line joining each point in the decision space to the origin (i.e. $\Psi_i = B_i/E_i = [B_i - 0]/[E_i - 0]$). This has been shown in Figure 17 below, with Ψ_i and Ψ_j depicting the numeric eco-efficiency of any two alternatives *i* and *j*, respectively.

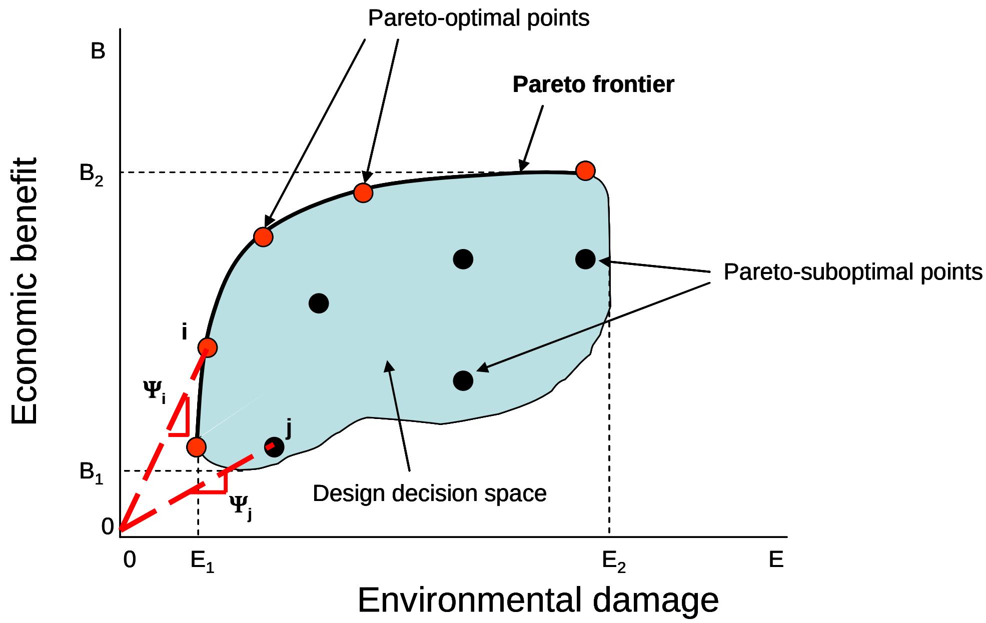

Figure 17: Illustration of a 2-dimensional design decision space and its relationship to eco-efficiency indicators

Having presented this alternative method of interpreting eco-efficiency, it is now of interest to relate it to the tactical and operational design decision contexts that are encountered in minerals process design and are of interest in this thesis. These relationships are discussed next.

3.1.2.2 Application of eco-efficiency to tactical design decision contexts

In most commercial minerals beneficiation projects, a positive economic benefit (typically measured in profit or return on investment) is desired. However, these projects are inevitably associated with a certain amount of environmental burden. At tactical levels of process design, this (positive) economic benefit needs to be weighed against the environmental impact of each considered process during decision making towards the final design. In this design context, the adverse environmental impact is measured in a *positive* sense, i.e. the alternatives in the design space yield positive economic returns at the expense of a (positive) environmental damage (i.e. $\Psi > 0$ in Equation 2). The design space therefore resembles that in Figure 17 above, where a **high** and **positive** value of eco-efficiency is desired. Furthermore, as explained above, Pareto optimality dictates that design alternatives on the Pareto curve or 'decision frontier' be preferred over those that are sub-optimal (i.e. below the Pareto curve in Figure 17). The combination of these two conditions gives rise to different cases that the decision maker may face in choosing the most optimal process route to further develop in tactical design. These have been shown in Figure 18 below – a design space containing the economic and environmental performance values of a set of design alternatives *i*, *j*, *k*, *m*, *n* and *o*.

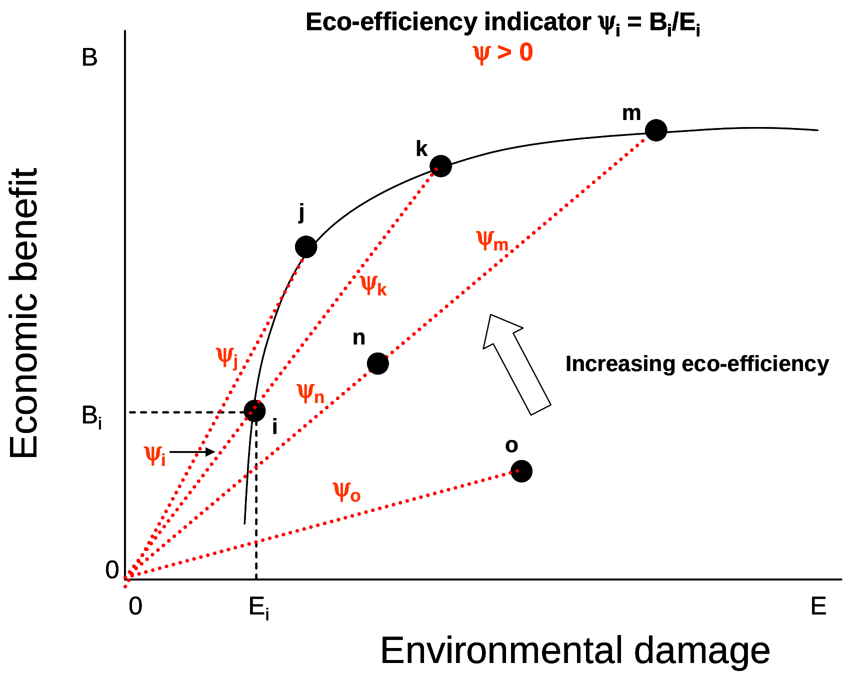

Figure 18: Graphical mapping of eco-efficiency onto a Pareto design space in a tactical design decision context

In the above figure, as Pareto-optimal points on the design space decision frontier, alternatives *i, j, k* and *m* are 'best-performing' and are thus preferable to alternatives *n* and *o.* When the numeric eco-efficiency of these alternatives is considered (i.e. Ψ values as slopes of the lines between each alternative and the origin), it can be observed that eco-efficiency improves as one moves from the sub-optimal region of the design space towards the decision frontier. Maximum eco-efficiency is achieved with the slope whose line to the origin is tangential to the Pareto curve (i.e. point *j*). A design alternative characterised by point *j* therefore represents the optimum process which yields the highest economic benefit at the least environmental impact. Should there be no such alternative in the set of process options, optimisation exercises on alternatives with performance values closest to this point (e.g. option *i* or option *k* above) could be performed.

Further interesting cases can be noted from the above theoretical analysis. It can be observed that the convexity of the Pareto curve in this case makes it possible for this frontier to be intersected more than once by the same line from the origin. This essentially means that it is possible for two or more design alternatives to have the same numeric eco-efficiency while characterised by different values of economic and environmental performance in the decision space (e.g. design options *m* and *n*). However, *no more than two* design alternatives with the same eco-efficiency can also be Pareto-optimal (e.g. design options *i* and *k*). The first observation is obvious when the ratio nature of eco-efficiency indicators is considered: theoretically, an infinite combination of values describing the numerator and denominator of an eco-efficiency indicator can yield the same ratio. However, the second is more obscure: for each value of eco-efficiency associated with these combinations, a maximum of only *two* combinations can yield Pareto-optimal solutions. These theoretical deductions may be useful when incorporated into computational algorithms (such as the computational design methods mentioned in Chapter 2) that optimise a vast number of design alternatives to arrive at optimal solutions during the evolution of the base case design.

3.1.2.3 Application of eco-efficiency to operational design decision contexts

As mentioned in Chapter 2, the majority of operational design decisions are concerned primarily with *technical* performance improvement, such as in retrofit design. In many cases, all technical improvements achievable are associated with a net financial cost to the overall process, i.e. an economic loss[44]. In Figure 17 above, this implies that B < 0 for any design alternative *i*. Figure 19 below illustrates this case, for a set of design alternatives *w, x, y* and *z*.

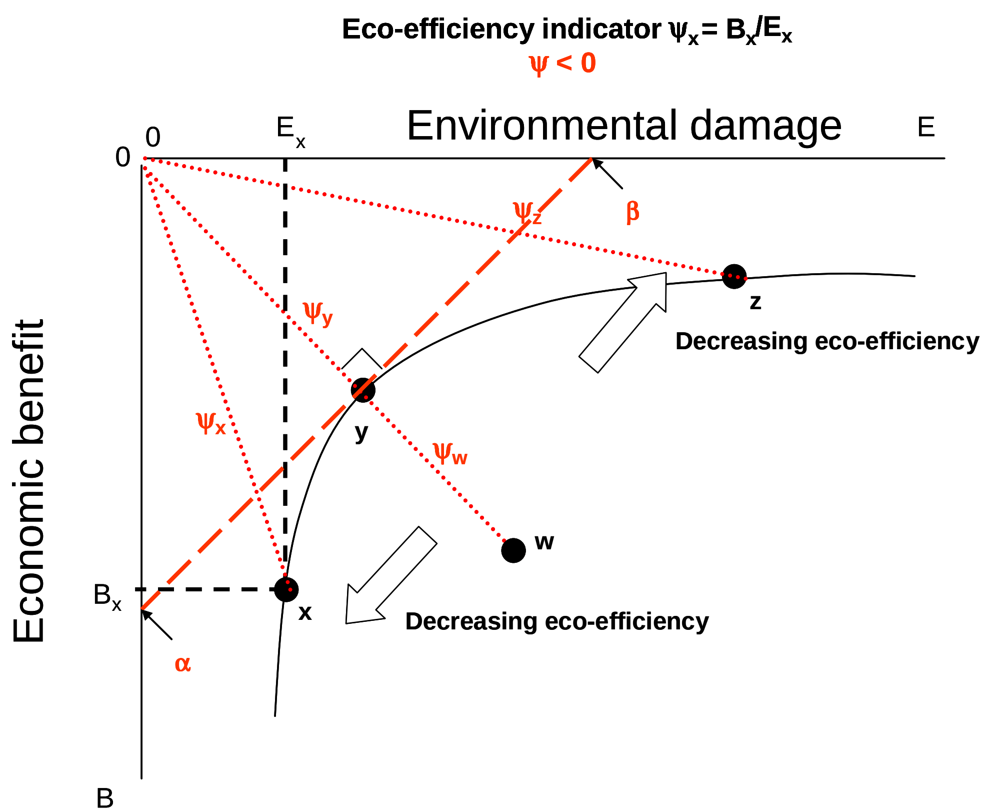

Figure 19: Graphical mapping of eco-efficiency onto a Pareto design space in an operational design decision context

Since B < 0, the eco-efficiency indicator Ψ_i for each alternative now also assumes negative values. As the economic loss is to be limited, a **low** and **negative** value of eco-efficiency is

[44] Whilst there usually is a good justification for such improvements, the benefits (e.g. safer operation) may be 'intangible' or not-quantifiable for the immediate decision context, and are thus typically set to zero (von Blottnitz, 2008).

now desired; without, however, extending environmental damage too far. Since both the economic loss and the environmental damage now need to be minimised, the optimal design alternative is characterised by the point on the Pareto curve that is closest to the origin – a certain point *y* in Figure 19. The eco-efficiency of this optimum alternative is therefore Ψ_y, the slope of the line between *y* and the origin. However, geometrically, the distance between the origin and point *y* is at its minimum when Ψ_y is perpendicular to the line α–β that is tangential to the decision frontier at point *y*, i.e. the product of Ψ_y and the slope of line α–β is equal to -1. This derivation can be used in this design context for determining the optimal design alternative or operating point for achieving 'maximum' eco-efficiency.

The above theoretical analysis has demonstrated that in those operational design decision contexts where economic loss is experienced, *distance* from the origin rather than the *slope* of the line from the origin to each design alternative is the governing criterion for optimising the eco-efficiency of the design alternatives. This shows that a meaningful interpretation of eco-efficiency needs to take into account the decision context in which the design procedure is carried out. According to this analysis, in some operational design decision contexts, the numeric eco-efficiency indicator alone is insufficient in guiding the selection and design of more environmentally sustainable processes; the environmental and economic performance values need to be directly considered (rather than in an eco-efficiency ratio). Further evidence of this deduction can be inferred from the need to consider limits to the acceptable levels of environmental damage *during* decision making, as opposed to being exogenous to this procedure: unless the 'too far' value of environmental damage mentioned above is known (i.e. the upper bound on the acceptable environmental damage for the design problem being investigated), the condition that Pareto-optimal points are preferable to sub-optimal points for *all* possible levels of environmental damage cannot be assumed as satisfied (e.g. in Figure 19, it cannot be confidently stated that alternative *z* is preferable to alternative *w* unless it is known whether the environmental impact associated with *z* exceeds a certain acceptable maximum). Another obvious observation that sharpens the differences between these decision contexts as constructs in this thesis is the number of possible Pareto-optimal points that can be described by the same eco-efficiency values: given that in the operational design decision context the lines joining each design alternative to the origin can only intersect the Pareto curve once, each Pareto-optimal design alternative can only be associated with *one* eco-efficiency value (as opposed to up to two values for the tactical design decision context). This theoretical critique therefore makes a solid case for the need to interpret eco-efficiency according to the process design decision context in which the analysis is performed.

3.1.2.4 Implications of eco-efficiency performance representations on distinguishability

While the above discussion has highlighted the difference in the interpretation of eco-efficiency between the tactical and operational design decision contexts, it has ultimately been based on the assumption of continuity associated with Pareto optimality, i.e. the existence of a Pareto curve or a continuous decision frontier. This may not necessarily be applicable for all minerals process decision contexts; indeed, many design problems in the minerals and primary metals industries are poorly defined or characterised by discrete technologies as design alternatives that make it difficult to establish continuous regions of performance from which Pareto curves can be inferred (Stewart, 1999). This 'discontinuity' can be mitigated through the use of 'relative' eco-efficiency as an alternative method of graphically depicting the environmental and economic performance of design alternatives without the inherent continuity assumption of Pareto optimality. In this representation, the environmental (E) and economic (B) performance values are 'normalised' with respect to the total contribution for all the considered alternatives. The methodology for achieving this is shown through Equation 3 below, using economic benefit as an example.

$$b_i = \frac{B_i}{\sum_{k=1}^{n} B_k} \cdot n$$

Equation 3: Normalisation formula for computing relative eco-efficiency

Normalised economic and environmental performance values can then be plotted on an XY-diagram that can be divided into four quadrants. Conventionally, quadrants are plotted such that options with the highest eco-efficiency are found in the top right quadrant (i.e. quadrant II), with those exhibiting the lowest eco-efficiency in the bottom left quadrant (i.e. quadrant IV), as shown in Figure 20 (Michelsen *et. al.*, 2006; Saling *et. al.*, 2002). The Cartesian coordinates (1,1) then represent a reference case to which the eco-efficiency of all design alternatives may be compared. This reference case may be arbitrary (e.g. Saling *et. al.*, 2002) or may be an actual design base case that already exists against which other alternatives may be compared.

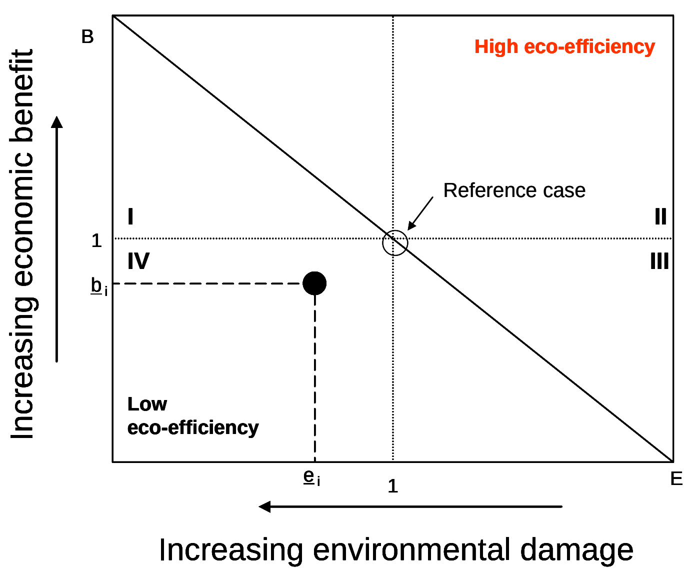

Figure 20: A normalised eco-efficiency XY-quadrant diagram

A key advantage of this approach is its ability to depict the performance of design alternatives using relative positive values that eliminate the complications associated with the operational design decision context as described in the preceding section. However, the approach is oversimplified in that it masks an important consideration that applies to all decision contexts and all representations of environmental and economic performance representations: to what extent are the comparisons of the environmental and economic performance of the design alternatives meaningful when uncertainty is taken into account? This section therefore seeks to explore this consideration.

In Chapter 2, it has been demonstrated from the literature that the extent to which the design alternatives can be distinguished (i.e. the extent of overlap in the performance values or ranges of the design alternatives) is a critical consideration in decision making during process design. It is therefore useful to now extend the 'decision space' concept introduced in Chapter 2 to explicitly account for uncertainty. Basson (2004) states that values characterising performances of design alternatives in both conceptual (tactical) and detailed (operational) design decision contexts need to be communicated together with their levels of uncertainty. These decision spaces have been shown once more in Figure 21(a) and Figure 21(b) for the tactical and operational design decision contexts, respectively.

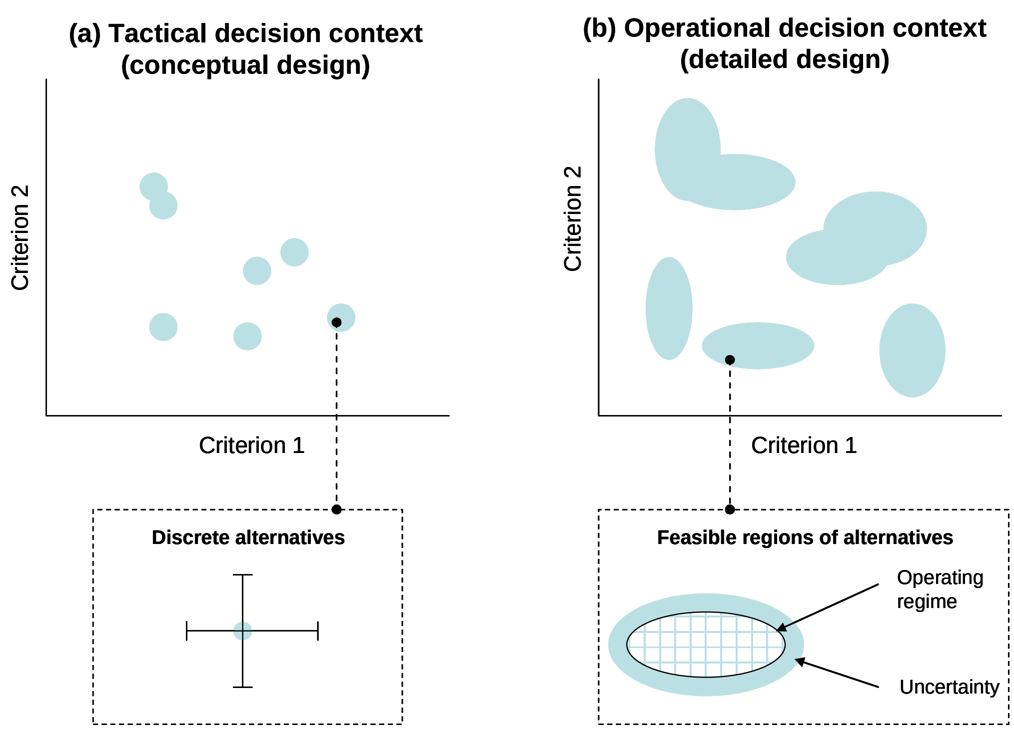

Figure 21: The tactical and operational design decision contexts, with their associated levels of uncertainty[45]

[45] After Basson (2004)

It is of interest in this thesis to consider how eco-efficiency indicators, when compared to the graphical approaches presented above, communicate the performance of design alternatives *in different decision contexts*. A key argument of this thesis is that distinguishability (which in turn is dependent on uncertainty characterising the performance values or ranges of the design alternatives) will exert a significant influence over the decision objectives to which the eco-efficiency indicators need to respond. This contention supports the claims made in section 3.1.2.2 and section 3.1.2.3 above, which call for the need to explicitly consider the decision context when applying eco-efficiency to process design. The second hypothesis for this research can therefore be stated as:

"The use of eco-efficiency indicators to inform decision making in minerals process design needs to be governed by the decision context."

(Hypothesis 2)

3.1.3 Statement of the research hypothesis

The above discussion has affirmed that the context in which process design decisions are made needs to be understood so as to elucidate approaches, tools and methodologies which can yield more environmentally sustainable projects within the minerals industry.

Having made a case for this research in section 3.1.1and section 3.1.2 above based on observations from the literature, two parts of the hypothesis that is to be tested in this thesis were formulated. These can be combined as follows:

Eco-efficiency indicators can meaningfully communicate the environmental and economic performance of design alternatives in minerals process design. However, their use as sustainability performance metrics to inform decision making needs to be governed by the decision context; i.e. the performance indicators need to be *fit-for-purpose*.

The research approach and methodology to interrogate the above hypothesis is presented in the following section.

3.2 Research design

3.2.1 Research approach

The hypothesis stated above will be tested in this thesis through case studies. A rationale for the use of this research approach is offered in this section.

Case study research is a well-established research design that is used to provide an in-depth description of a small number of cases (Mouton, 2001). Yin (1994) defines a case study as *"an empirical enquiry that investigates a contemporary phenomenon within its real-life context, especially when boundaries between phenomenon and context are not clearly evident".* Emphasis on case study research has been on analysing real-life situations, in an array of contexts: companies or organisations, engineering, social studies and political sciences (Mouton, 2001). However, across these different fields, the most prevalent application of case study research has been towards *theory-building*, i.e. the production of new knowledge from novel theory (Woodside and Wilson, 2003; Mouton, 2001). In-depth case studies have been successfully used to produce new insights, notably in the areas of operations research and management sciences (e.g. Jaspers, 2007; Wacker, 1998). Furthermore, there is recent evidence for their use in the process systems engineering literature for developing environmental decision making theory in resource-based industries (e.g. Broadhurst, 2007a; Giurco, 2005; Basson, 2004; Notten, 2002; Stewart, 1999). These contributions set an important precedent for the use of case studies in this thesis, as motivated below.

In Chapter 2, it has been highlighted that there currently exists a knowledge gap in the integration of environmental considerations into minerals process design. Particularly, section 3.1 above has demonstrated that while eco-efficiency shows value in simultaneously communicating the environmental and economic performance of design alternatives to decision makers in the minerals beneficiation community, there is a need to understand the relationship between the eco-efficiency metrics as sustainability indicators and the decision context in which they are used for meaningful decision making to be achieved. When related to Yin's definition above of the case study, eco-efficiency indicators therefore constitute the "phenomenon" or construct that is of interest in this thesis, while the "context" can be explicitly related to decision making during minerals process design. The case study approach is therefore suitable for mapping the linkage between eco-efficiency indicators and the decision context for the purpose of theory building. As such, it is envisaged that using case studies in this thesis to derive in-depth insights that fill the knowledge gap between environmental decision making theory and minerals process design will make a useful contribution to the minerals beneficiation community.

3.2.2 Motivation and selection of case studies

In testing the stated hypothesis, the case studies employed in this thesis were selected to reflect different decision contexts that are typically encountered in minerals process design, whereby engineers and technical specialists are key decision makers. The first case study represents a tactical design decision context, where eco-efficiency is applied for the economic and environmental performance assessment of a suite of copper beneficiation process technologies. In the second case study, eco-efficiency is used as an environmental performance analysis tool for a retrofit design of a tailings dewatering circuit in a gold processing facility. Drawing on descriptions and guidelines offered by Notten (2002) and Basson and Petrie (2001), details on the selection of these case studies are provided below.

3.2.2.1 The tactical design decision context: Case study 1

The tactical design case study represents decision situations encountered during the conceptual (early) stages of process design, where a 'new' or greenfield mineral beneficiation project is being considered. The process alternatives studied are characterised by relatively broad system boundaries incorporating the entire minerals-to-metals value chain as well as a high degree of uncertainty in the process data. The primary decision objective is therefore *preliminary technology selection*, i.e. this phase of the design procedure is considered complete when a primary technological process route is selected for further development into a full process flowsheet.

The broad system boundaries and low resolution of process information necessitate the consideration of *all* relevant environmental impacts associated with each potential processing route. Input-output or block flow diagrams are used to depict the various processing routes. While the number of process alternatives considered can theoretically be infinite (Basson, 2004), *a priori* knowledge and case-based reasoning is typically used to frame these as a finite number of possible technology options (drawing on information such as market conditions, industry best practice, ore body characteristics etc.). Since simple process models are used to generate material and energy balance information, it is appropriate to calculate performance ranges based on design heuristics only.

3.2.2.2 The operational design decision context: Case study 2

In the operational design context, decision situations are encountered during the more detailed (later) stages of process design, and can often be associated with existing operations. The process alternatives studied are characterised by relatively narrow system boundaries enclosing only a certain section or circuit of the entire operation; they are also described by good quality (i.e. well-defined) process data. The primary decision objective is therefore *performance improvement* in a constrained environment, where a new and specific need arises within the operation and drives the design procedure (e.g. a change in the fundamental properties of the treated ore body, changes in resource availability, new regulations or policies etc.).

Due to the narrow system boundaries and high resolution of process information, only the environmental impact(s) that are tied to the decision objective need to be considered. It is possible to describe the design alternatives with (more detailed) process flow diagrams at this phase, where the available options are well-known (even though their generation can still be on a case-by-case basis). Detailed process models are necessary to generate material and energy balance information, which in turn allows for more elaborate uncertainty analyses to be performed (e.g. sensitivity analyses).

The overall boundaries for the case studies have been shown in Figure 22 below (processes that are not shaded are included in the corresponding case study analysis).

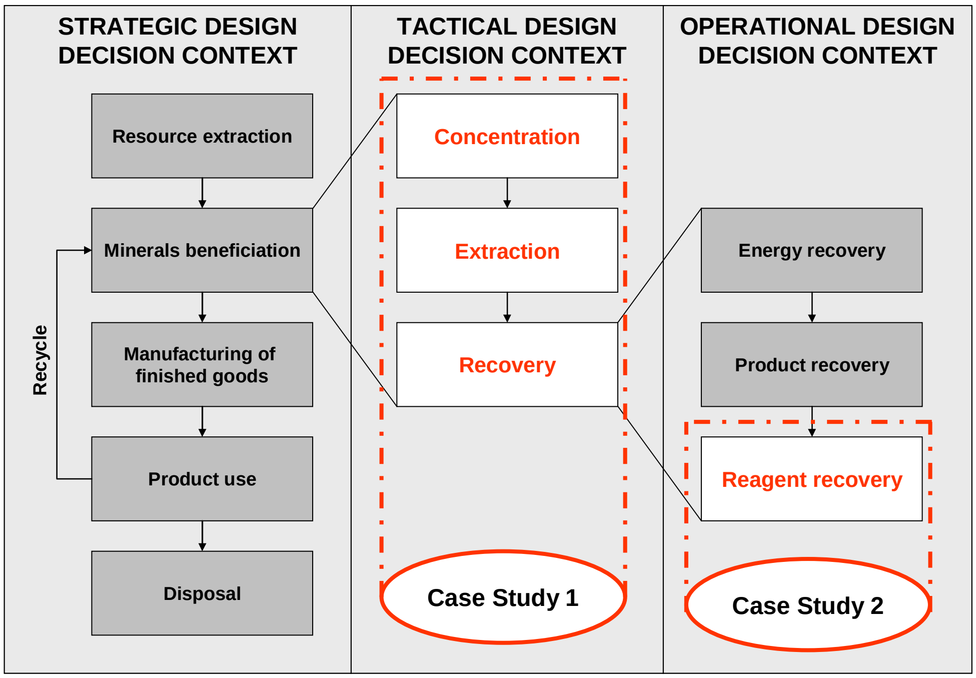

Figure 22: Case study boundaries

Descriptions of these two case studies are summarised in Table 5 to elucidate key differences between them.

Table 5: Summary descriptions of the tactical and operational design case studies

Process design step	Specification	Decision criteria (after Notten, 2002; Basson and Petrie, 2001)
TACTICAL DESIGN: CASE STUDY 1		
Problem framing	Decision objective	• Technology selection
	Choice of environmental performance indicators	• Life cycle impact assessment mid-point indicators
Generation of alternatives	Specification of design alternatives	• Block flow diagram
	Number of discrete alternatives	• Three[46]
Analysis of alternatives	Level of process model detail	• Simple models e.g. preliminary mass and energy balances
	Management of uncertainty	• Establishing likely ranges of performance indicators to guide screening
OPERATIONAL DESIGN: CASE STUDY 2		
Problem framing	Decision objective	• Process change
	Choice of environmental performance indicators	• Site-specific indicators and/or emissions-based criteria
Generation of alternatives	Specification of design alternatives	• Process flow diagram
	Number of discrete alternatives	• Six[47]
Analysis of alternatives	Level of process model detail	• Full material and energy balances of life cycle (only if earlier analysis shows that the environmental impact of the material value chain is sensitive to design variables), otherwise • Full material and energy balances of process only
	Management of uncertainty	Sensitivity analyses, including: • Parametric uncertainties • Model uncertainties

3.3 Research methodology

3.3.1 Specification of eco-efficiency indicators for the minerals industry

As mentioned in Chapter 2 and motivated in section 3.1.1 in this chapter, the use of eco-efficiency indicators in the minerals and primary metals industries still requires further development. This thesis will build on the contribution of van Berkel and Narayanaswamy (2005) to relate environmental impacts that were identified as of critical importance in the minerals beneficiation sector to eco-efficiency indicators as specifically defined in Chapter 1. The objectives for mitigating typical environmental impacts in minerals processing have been defined as:

1) Effective resource utilisation and materials efficiency,
2) Reduction of process residues and the enhancement of co-product values,
3) Reduction of water use and impacts,
4) Reduction of energy consumption and consequent greenhouse gas emissions, and

[46] In theory the number of discrete alternatives is infinite, but is often framed as a number of finite technologies with estimates for predicted performance (Basson and Petrie, 2001)
[47] Typically 3 – 5 alternatives are considered (Basson and Petrie, 2001)

5) Improved control of minor elements and toxic dispersion.

Van Berkel and Narayanaswamy linked each of these environmental objectives (with the exception of the control of minor elements) to **key performance indicators** (KPI's) as quantitative measure or eco-efficiency. These KPI's have been shown in Table 6 below, using an aluminium refinery as an example:

Table 6: Eco-efficiency themes, underlying issues and performance indicators in minerals processing[48]

Eco-efficiency theme	Project-specific Issues	KPI
Effective resource utilisation and materials efficiency	Aluminium recovery	% Recovery of alumina
Reduction of process residues and the enhancement of co-product values	Residue generation	ton residue/ton alumina
Reduction of water use and impacts	Water consumption Leachate generation from residue area	kL H_2O/ton alumina Sodicity of leachate generation
Reduction of energy consumption and consequent greenhouse gas emissions	Total energy consumption Total GHG emissions	GJ/ton alumina kg CO_2/ton alumina
Improved control of minor elements and toxic dispersion	NORM control	

While these KPI's embrace the eco-efficiency concept as they quantitatively integrate the environmental and economic performance aspects of a minerals beneficiation process simultaneously, there are five shortcomings that need to be addressed, as described below.

3.3.1.1 Eco-efficiency vs. Eco-intensity

An inspection of the KPI's defined above shows that these indicators are actually defined as *intensity* indicators rather than *efficiency* indicators (i.e. intensity of resource consumption or waste generation per unit of economic benefit (ISO, 1999), rather than the efficiency of resource utilisation relative to the associated adverse environmental impact (WBCSD, 2000)). While these are merely inverse forms of each other, for consistency the efficiency form of these indicators will be used in this work.

3.3.1.2 Environmental risks from minor toxic elements: Design for control vs. Design for elimination

It has been well accepted within the process industries that the presence of trace species in a process plays a crucial role in influencing system dynamics and thus overall system-wide controllability (Wu *et. al.*, 2002; Dimian *et. al.*, 1997). Owing to the often very complex mineralogy of the feed ore with many minor toxic elements as components, primary metals industries are no exception. The importance of the ore mineralogy becomes particularly important due to the often highly toxic nature of some of the minor elements in the ore,

[48] Source: Modified from van Berkel and Narayanaswamy (2005)

despite their low abundance[49]. Since at early stages of process design the need to minimise feasibility study costs often does not warrant detailed mineralogical analyses of the ore, control systems are therefore typically defined only once the base flowsheet has been set, rather than being developed *in tandem* with the production system. While other factors such as mineralogy may account for the lack of a generic KPI for the control of minor and toxic elements, this limited development of the control philosophy during process design is also envisaged to be a contributing factor. This implies that alternatives with attractive control advantages may be overlooked during flowsheet development e.g. alternatives which consume unwanted minor and trace elements. Furthermore, the control autonomy of a unit operation within a flowsheet may have important systemic effects on the overall performance of a flowsheet e.g. the effect of the presence or absence of a surge tank on the reactor and separator sizes and heat duties (and therefore capital and operating costs) in a reactor-separator-recycle loop.

The above observations therefore call for the adoption of a more systems-based approach through which opportunities to eliminate minor and trace elements as long-term environmental risks can be brought to light and potentially be harnessed i.e. ‘design for risk alleviation’ rather than ‘design for risk control’. This in turn encourages placing emphasis on a fundamental understanding of the origin and constitution of these minor elements in the ore body, as well as their behaviour and distribution throughout the process. There is increasing evidence of a shift towards this realisation in the literature, as evident in the use of thermodynamic semi-empirical and empirical models towards this end (Broadhurst, 2007a; Georgalli *et. al.*, 2002).

3.3.1.3 Eco-efficiency and value creation

In addition to overlooking opportunities for eliminating the environmental and health risks associated with minor toxic elements, the current design approach may overlook opportunities for additional value creation. For example, despite their low crustal abundance, many elements undergo significant enrichment in tailings disposal facilities over the typically long lifetimes of minerals beneficiation operations. As an example, Table 7 below shows typical

[49] Another major problem is that of large-volume solid wastes associated with this industry (Broadhurst, 2007a; IIED, 2001). It is argued here, however, that the fact remains that owing to the highly disseminated and low concentrations of valuable metals in gangue material, the generation of these wastes will unfortunately be an ugly hallmark of this industry for some time to come. Of course, minimising the generation of these wastes is a pressing concern; however, more emphasis should be placed on *understanding* the elemental deportment of the minor toxic elements present in this waste that are the real environmental concern (e.g. Broadhurst, 2007a), rather than the monitoring of the total amount of waste generated. This implies that environmental performance indicators should be designed to *directly* reflect the environmental risk associated with the minor toxic elements present in the waste, rather than using the total amount of waste generated as a ‘proxy’ for the real environmental impact. It is for this reason that the eco-toxicity eco-efficiency indicator is specified in this thesis, rather than a generic ‘value created per tonne waste’ indicator as suggested by the KPI in Table 6.

concentration ranges and enrichment factors[50] for major and selected minor elements in a porphyry-type Run-of-Mine (ROM) copper sulphide ore:

Table 7: Porphyry-type ROM CuS ore typical element concentration ranges & enrichment factors[51]

Element	Concentration range	Enrichment factor	Element	Concentration range	Enrichment factor
Major sulphide elements (%)			***Major lithophilic gangue elements (%)***		
Cu	0.5-1.0	100-200	Si	21-34	≤ 1.2
Fe	1-10	≤ 2	Al	4-10	≤ 1.2
S	2-11	50-200	Mg	0.2-3	≤ 1.2
			Ca	0.4-4	≤ 1.2
			K	0.3-3.4	≤ 1.2
			Na	0.3-3	≤ 1.2
Trace-minor sulphide elements (ppm)			***Trace-minor lithophilic gangue elements (ppm)***		
As	5-1800	10-1000	Ti	440-8800	≤ 2
Zn	150-1600	2-20	P	100-6000	0.1-5
Mo	15-1500	10-1000	F	60-3000	0.1-5
Pb	30-300	2-20	Mn	100-2000	≤ 2
Cd	2-200	10-1000	B	50-1000	5-100
Bi	2-200	10-1000	Ba	40-860	≤ 2
Sb	2-200	10-1000	REE	10-850	0.1-10
Ni	8-150	≤ 2	Rb	10-600	0.1-5
Se	10-100	100-1000	Sr	30-600	≤ 2
Ag	1.0-70	10-1000	Cl	10-500	0.1-5
Co	2.5-50	≤ 2	Zr	10-500	0.1-5
Ge	2-20	2-20	Li	5-300	0.1-10
Tl	0.6-6	2-10	Sn	15-300	5-100
Pt	0.05-5	10-1000	V	15-300	≤ 2
Au	0.04-4	10-1000	Cr	10-200	≤ 2
Pd	0.1-2	10-1000	Nb	2-200	0.1-10
Hg	0.2-1.5	2-20	W	5-100	5-100
Te	0.1-1	100-1000	Ga	2-80	0.1-5
In	0.1-1	2-20	Sc	1-70	0.1-5
Re	0.01-1	10-1000	Be	0.5-30	0.1-10
			Br	0.5-25	0.1-10
			Hf	0.5-25	0.1-10
			U	<1-10	0.1-5
			I	<1-5	0.1-10

It may therefore be possible that after a certain amount of time some of these elements may have been enriched to a concentration at which it may be economically and technically viable to beneficiate them from these wastes. A further benefit may be extracted if it is environmentally desirable to recover these elements and prevent their long-term release into the environment e.g. recovering highly toxic but highly valuable metallic elements with attractive market potential. The ability of eco-efficiency to identify improvements that are both

[50] Defined as the extent to which (or the number of times that) the concentration of a trace substance is enriched in a tailings impoundment
[51] Source: Broadhurst. (2007a)

economically desirable *and* environmentally responsible becomes evident in this case. Recovering wastes as by-products thus improves the resource efficiency of the process. While this opportunity is captured in the KPI's proposed by van Berkel (reduction of residue generated and the enhancement of co-product values), the consideration of residue in this aggregated form may be misleading from an economic perspective (e.g. if no trace elements are enriched enough or technologically recoverable). A simple screening methodology similar to that offered by Broadhurst (2007a) would be appropriate in adequately describing the potential of a process for by-product value creation.

3.3.1.4 Economic cost vs. Economic value added

In the context of eco-efficiency, economic benefit is interpreted as additional value which a product, process or service contributes to society. Thus, by definition, the numerator of any eco-efficiency indicator should include both the financial return *and* cost associated with the creation of that product or service. For the case of primary metals production, this observation implies that both the gross revenue *and* unit production costs need to be considered. In the above KPI's, only the income (from product sales) is used to characterise the numerator of the eco-efficiency indicator. In comparative analyses (as often is the case at process design when comparing the techno-economic and environmental performance of various process alternatives), such an omission may lead to severe distortions of the values of the eco-efficiency indicators and consequent gross misrepresentation of the true performance ranking within the set of design alternatives.

3.3.1.5 Life cycle assessment-based environmental impacts

There is an increasing awareness of the need to quantify environmental impacts from a life cycle perspective in the minerals and primary metals industries, in order to fully account for all impacts and avoid designing processes or implementing interventions that merely shift the environmental burden up or down the minerals-to-metals value chain (Petrie, 2007; Stewart *et. al.*, 2003; Azapagic, 2003). In keeping with this practice, the eco-efficiency themes relevant to the minerals and primary metals industries have been linked to well-accepted life cycle environmental impacts and their associated indicators in Table 8 below.

Table 8: Life cycle impact assessment indicators linked to eco-efficiency themes[52]

Eco-efficiency theme	Life cycle environmental impact	Life cycle impact assessment indicator
Effective resource utilisation and materials efficiency	Resource depletion	kg antimony-equivalents
Reduction of process residues and the enhancement of co-product values		
Reduction of water use and impacts	Dissipative water use	m^3 water
Reduction of energy consumption and consequent greenhouse gas emissions	Global warming	kg carbon dioxide-equivalents
Improved control of minor elements and toxic dispersion	Eco-toxicity	1,4 dichlorobenzene-equivalents

3.3.1.6 A modified set of eco-efficiency indicators proposed

Based on the above 5-point critique a revised set of eco-efficiency indicators are proposed in Table 9 below.

Table 9: Proposed eco-efficiency indicators for the minerals beneficiation sector

Eco-efficiency theme	Eco-efficiency indicator	Indicator units
Effective resource utilisation and materials efficiency **(Resource depletion)**	**Major/primary product resource efficiency:** Primary product value created per unit of valuable mineralogical material extracted	US$/tonne Sb-equivalents
	Minor/by-product resource efficiency: By-product value created per unit of valuable mineralogical material extracted	US$/tonne Sb-equivalents
Reduction of water use and impacts **(Dissipative water use)**	Value created per unit of water dissipated	US$/$m^3$ H_2O dissipated
Reduction of energy consumption and consequent greenhouse gas emissions **(Global warming)**	Value created per unit of greenhouse gases emitted	US$/kg CO_2-eq. released
Improved control of minor elements and toxic dispersion **(Aquatic eco-toxicity)**	Value created per unit of toxic elements released	US$/tonne 1,4-dichlorobenzene-equivalents released

3.3.2 Overall approach for the computation of eco-efficiency

The computation of eco-efficiency indicators was based on quantifying the environmental and economic performance of the design alternatives considered in each case study. Economic performance was determined as the difference between the cash revenue generated from

[52] After ISO (1999)

metal production sales and the annualised capital and operating costs estimated for each alternative (in US$). Material balance data were used to determine the environmental impacts associated with each alternative for the resource depletion (tonne Sb-eq), dissipative water use (m^3 H_2O dissipated), global warming (kg CO_2-eq.) and eco-toxicity (tonne 1,4 dichlorobenzene-eq.) environmental impact categories. For each environmental impact category, eco-efficiency indicators were computed as the ratio of the economic performance and the corresponding environmental performance. The eco-efficiency indicators were then compared to the graphical representations of economic-environmental performance that were described in sections 3.1.2.2 and 3.1.2.3. Sensitivity analyses were also conducted to identify the influence of various process design parameters on the eco-efficiency performance of the design alternatives. A simple conceptual scheme depicting the overall approach in computing the eco-efficiency indicators is shown in Figure 23.

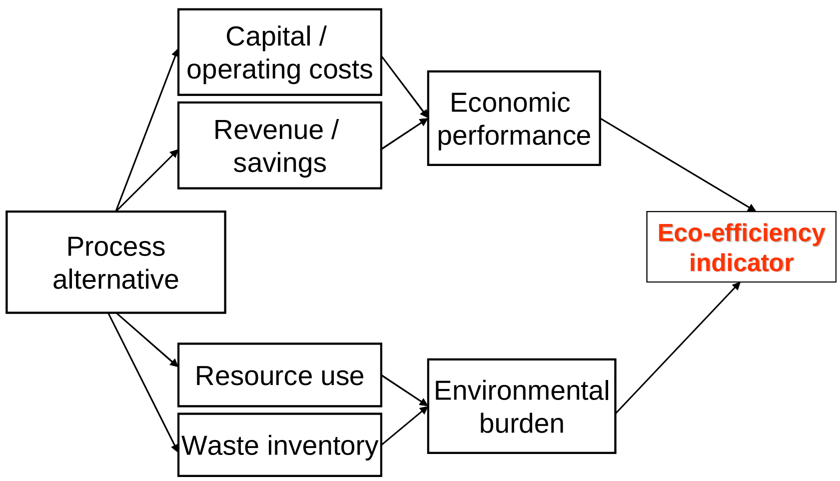

Figure 23: Simple conceptual scheme for computing eco-efficiency indicators

Detailed accounts of the methodologies adopted are included in Chapter 4 and Chapter 5, alongside each case study. Results from each of these case studies are presented in Chapter 4 and Chapter 5.

3.3.3 Distinguishability analysis

In efforts to contribute towards a clearer elucidation of environmental performance information for decision making during process design, Basson (2004) proposed a 'distinguishability index' approach to provide guidance on performance information in determining:

1) whether a comparative evaluation of process design alternatives can be carried out meaningfully despite uncertainty in their performance values, and
2) which uncertainties can be targeted for reduction, and to what extent this can be done to make the alternatives distinguishable from one another.

The methodology developed by Basson is based on using performance information and the associated uncertainty data to compute dimensionless indices that are a measure of the extent to which the performance information characterising process design alternatives can be distinguishable from one another in the design space. This is performed in a pair-wise manner for all design alternatives considered i.e. each design alternative is separately analysed for distinguishability from each of the other alternatives in the design space in turn. This distinguishability index approach is used in this thesis as described below.

The calculation of the distinguishability index for each design alternative requires three key sets of data:

1) the *best guess* of a performance value for a certain performance criterion,
2) a *likely maximum* of the performance value, and
3) a *likely minimum* value for the performance scores of each alternative for each performance criterion

The difference between the best guess and the likely maximum and that between the best guess and the likely minimum form the *positive dispersion threshold* and *negative dispersion threshold* respectively (i.e. the positive and negative uncertainty ranges associated with the best guess). A simple scheme of this analysis is shown in Figure 24 below.

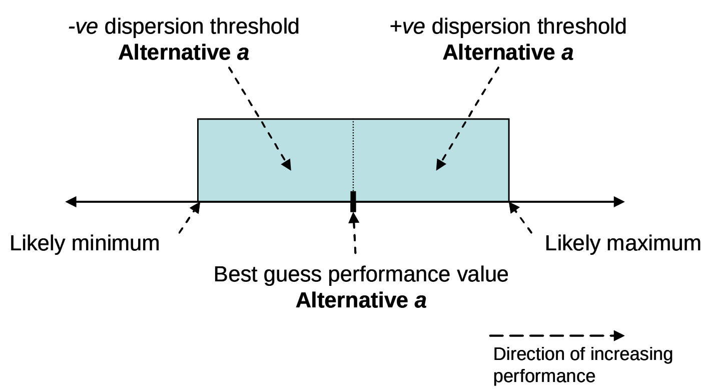

Figure 24: Illustration of best-guess performance value and distinguishability thresholds for a process alternative

To determine whether the alternatives are distinguishable, two sets of information are determined:

1) the difference between the best estimates (i.e. ‘best guesses’) of the performance values for each performance criterion, calculated for all pairs of alternatives based on process data, called *best estimate differences*, and
2) the **required minimum** difference between the best estimates of the performance values of two alternatives (e.g. *a* and *b*) for a particular performance criterion which

ensures that the alternatives are distinguishable from one another, referred to as the *distinguishability threshold*

The alternatives can be compared in a pairwise manner for each criterion to determine whether they are distinguishable from each other. An alternative is therefore defined as completely distinguishable from another when the best estimate difference far exceeds the distinguishability threshold, and completely indistinguishable when the best estimate difference is much less than the distinguishability threshold. In quantifying this assessment, a value can be assigned to a distinguishability parameter to specify whether two alternatives are distinguishable from each other when considering a particular performance criterion. Distinguishability between two alternatives can be indicated by a number 1, and indistinguishability by a value of 0. Since an indication is required of whether the alternatives are distinguishable from each other considering all the performance criteria, the information regarding distinguishability can be aggregated across the criteria into a distinguishability index (DI) for each pairwise comparison of alternatives The values for the DI can then be aggregated across all the pairwise comparisons into a single score for each of the alternatives considered, the aggregated distinguishability index (ADI). A value of 0 for the ADI of a certain alternative implies complete indistinguishability from all other alternatives, while a value of 1 indicates that an alternative is completely distinguishable from other alternatives. Intermediate values imply ‘weak’ or ‘partial’ distinguishability. As recommended by Basson (2004), an ADI value of 0.5 is used as a ‘cut-off’ value between weak and partial distinguishability. The logic of the distinguishability index approach and more detailed illustrations are provided in Appendix C.1.

3.4 Summary

Chapter 3 has expanded on the eco-efficiency literature review that was offered in Chapter 2 to make a research case (as represented in two hypotheses) for the application of eco-efficiency to the minerals industry in general as well as in both tactical and operational process design contexts within this sector. A motivation for a case-study research approach was then offered, with two case studies chosen to reflect each of the tactical and operational design contexts typically encountered in minerals beneficiation. Further detail on the selection of eco-efficiency indicators that are the most relevant to minerals process design and the overall methodology for their computation was also provided. Finally, a methodology to test the ‘meaningfulness’ of these eco-efficiency indicators relative to other approaches to eco-efficiency performance was outlined.

The research hypotheses, design and methodology that were developed in this chapter are then applied to the selected case studies in Chapter 4 and Chapter 5 of this thesis.

CHAPTER 4

Case Study 1: Eco-efficiency and the Tactical Design Decision Context

This chapter presents findings from the first of the two case studies investigated in this thesis. This case study interrogates the application of eco-efficiency for the economic and environmental performance assessment of copper beneficiation processes during process design. This application is considered at a *tactical* level of decision making, as explained in Chapter 3, where a preliminary screening of possible technologies available for processing is sought.

After describing copper beneficiation generally, as well as the major processing and beneficiation technology options that form the basis for the choice of design alternatives used in the case study, this chapter presents key observations on the eco-efficiency performance of the various alternatives in a tactical design decision context.

4.1 Background and description to the case study

4.1.1 Background to the copper beneficiation industry

Metallic copper has been known and mined by early man since around 7000 BC (Riekkola-Vanhanen, 1999). Copper is used extensively in electrical, electronic, industrial equipment and general consumer product applications, as well as in the construction and transportation industries (International Copper Study Group, 2008). While copper mineral deposits occur throughout the world, they are significantly concentrated in the western mountainous regions of North and South America (Biswas and Davenport, 1994). Mineralogical forms of copper are typically classified into sulphide, oxide or native (elemental) forms, with sulphide deposits of chalcopyrite ($CuFeS_2$) and bornite (Cu_5FeS_4), and oxide forms of cuprite (Cu_2O) and malachite ($CuCO_3.Cu(OH)_2$) being the dominant primary mineral ore types, respectively[53] (Giurco, 2005). Over 80% of all copper from mineral ores is recovered from sulphide ore deposits (Giurco, 2005). Primary global production[54] of refined copper in 2007 was recorded

[53] Important secondary mineral types include covellite (CuS) and chalcocite (Cu_2S), usually formed by the weathering and leaching of primary mineral deposits which re-precipitate near the groundwater table (Giurco, 2005). Refer to Broadhurst (2007a) for a detailed overview of the mineralogy of copper ores.

[54] "Primary" metallic copper refers to refined copper beneficiated from copper ores, while "secondary" copper describes copper produced from the recycling of scrap. While the latter is an important source of copper, accounting for approximately 40% of the copper reaching the market (Biswas and Davenport,

at 18.4 Mt (million metric tons) (International Copper Study Group, 2008), with an average spot price of approximately US$ 7,120 per metric tonne (Datastream, 2008). Production is forecast to increase to 20.9 Mt by 2009, driven primarily by market demand arising from China's high economic growth (International Copper Study Group, 2008).

An increasing awareness of the environmental impacts of primary metallic copper production has directed recent research efforts into improving understanding of these impacts to develop effective strategies for their mitigation towards environmental sustainability, in both local and global contexts (e.g. Broadhurst, 2007a; Giurco, 2005; Hansen, 2004). In particular, the large volumes of solid wastes, high resource consumption (including fuel, electricity and water) and airborne emissions and particulates associated with copper mining and beneficiation remain significant environmental concerns (Ayres *et. al.*, 2002). In line with the objectives of this thesis, it is therefore of interest in this case study to investigate whether eco-efficiency can meaningfully communicate the economic and environmental performance of copper beneficiation and processing technology options as process design alternatives in a tactical design decision context. These technology types and the models used to generate the eco-efficiency indicators are described in sections 4.1.2 and 4.2.1 below.

4.1.2 Description of copper beneficiation and processing technologies

Copper is beneficiated from ore deposits through two principal processing routes: *pyrometallurgical* and *hydrometallurgical* extraction. These are briefly described below.

Pyrometallurgical processing is based on the use of heat to oxidise the sulphur (and iron) usually present in concentrated sulphidic copper ores to produce a molten sulphide phase rich in copper (called the 'matte') which can be separated and recovered from a molten, copper-deficient oxide phase (called the 'slag'). However, due to the typically low mined copper ore grade – usually between 0.5% and 2% by weight (Biswas and Davenport, 1994), this process (known as 'smelting') is usually preceded by froth flotation to concentrate the copper and remove gangue material for disposal as tailings. The slag from smelting can be disposed of directly or can be further processed to remove impurities and/or recover any copper still present. The copper-rich matte is converted to molten 'blister' copper via the introduction of oxygen to remove iron and sulphur, and thereafter is refined to high-purity metallic copper (>99% Cu) through anode casting and electrorefining. The use of the pyrometallurgical processing route is related to the relative abundance of copper sulphide ores (compared to oxide ores), with more than 80% of all primary metallic copper produced from pyrometallurgical operations (Giurco, 2005). Pyrometallurgical processing options for copper are relatively mature and have been well-documented in the literature (e.g. Ayres *et.*

1994), the case study in this thesis is concerned with the former type of copper by source, i.e. copper extracted and processed from copper ores, as explained in section 4.1.2.

al., 2002; Riekkola-Vanhanen, 1999; Biswas and Davenport, 1994). Their technological development has been historically driven by the smelting stage of the process, with reverbaratory furnaces, electric furnaces, flash smelters and Noranda smelters as the four main types of smelting technologies in current use (Ayres *et. al.*, 2002). Flash and Noranda smelters are newer, more efficient continuous-operation technologies that are replacing the use of the older, less efficient batch-type reverbaratory and electric furnaces (Giurco *et. al.*, 2000).

As summarised by Giurco (2005), hydrometallurgical extraction of copper is currently achieved by percolating an acid solution through a crushed body of ore piled into a heap, then purifying and recovering the copper from the resultant pregnant liquor. Purification of the copper-rich acid leach solution is achieved using solvent extraction, after which the copper is electroplated from the solution (a process usually referred to as 'electrowinning') to recover the final product. This processing route is known as heap leach/solvent extraction/electrowinning (often abbreviated SX-EW or HL-SX-EW). Detailed descriptions and aspects of the HL-SX-EW processing route have been provided by Jergensen (1999). While this option currently accounts for less than 20% of all primary copper production, due to its flexibility in being able to treat both sulphidic and oxide ores[55] as well as its ability to accommodate low copper ore grades (< 0.25%), it represents the most rapidly expanding suite of technologies under development (Jenkins *et. al.*, 1999). Given the expected shift to the mining of predominantly low-grade sulphide ores and oxide ores in future as richer sulphide ore deposits get depleted, hydrometallurgical processing is deemed to be of key importance in future copper mineral development projects throughout the world (Jenkins *et. al.*, 1999). The recent development of biological heap leaching processes (more commonly known as 'bioleaching') into a minerals and metals extraction technology in its own right serves as a concrete example of this shift towards hydrometallurgical processing (e.g. Petersen and Dixon, 2006; Brierley and Brierley, 1999). Nonetheless, HL-SX-EW is still typically used to treat only low-grade copper ores (Norgate *et. al.*, 2007).

Flash smelting is the dominant technology type currently in use within the copper beneficiation industry (Giurco, 2005). In particular, the Outokumpu™ technology developed by the Finnish company Outotec accounts for more than 50% of all operational smelters globally alone, and continues to be the premium smelting technology of choice for new operations (Jenkins *et. al.*, 1999). However, reverbaratory furnaces still account for approximately 15% of worldwide pyrometallurgical copper processing capacity (Giurco, 2005). The use of electric

[55] It has been noted that the long leaching times required for satisfactory recovery of copper into solution from chalcopyrite ores (the most abundant copper-containing sulphidic ore type) often makes this option financially unattractive (Biswas and Davenport, 1994). However, in line with the underlying premise in this thesis for the need to go beyond the use of only techno-economic performance criteria such as the above for decision making in process design, this processing route is included in this case study analysis. The work of Giurco and colleagues (Giurco and Petrie, 2007; Giurco, 2005; Giurco *et. al.*, 2000) also sets a literature precedent for the argument made here.

furnaces for primary copper production is extremely limited, with most operational furnaces principally used for slag cleaning (Giurco, 2005; Biswas and Davenport, 1994). The envisaged future importance of hydrometallurgical operations such as the heap leach/solvent extraction/electrowinning has been described above. Based on this information, the technology options listed below will therefore used in this case study:

1) Reverbaratory smelting (abbrev. REVERB.),
2) Flash smelting (using Outokumpu^{TM} technology, abbrev. FLASH), and
3) Heap leach/solvent extraction/electrowinning (abbrev. HL-SX-EW).

The process flowsheet models used to calculate the eco-efficiency indicators for the above technology options (together with the underlying assumptions) and a detailed methodology are described in the following section.

4.2 Model development

4.2.1 Flowsheet specification and principal assumptions

The copper processing flowsheets used in this case study were based on a set of models constructed by Giurco (2005) to better understand the effect of primary copper processing technologies on the environmental burden associated with the global copper material value chain in its entirety. The technology options were modelled in a Microsoft EXCEL^{TM} environment, to a level of detail and accuracy consistent with the tactical design decision context of interest in the work presented in this thesis. Guirco (2005) was interested only in the environmental performance of the technology options was of interest in his work; this case study extends these models by introducing an economic dimension to the performance criteria for the considered technology options as process design alternatives. In this manner, the eco-efficiency performance for each alternative can thus be computed.

The flowsheets for the reverbaratory smelting, flash smelting and heap leach/solvent extraction/electrowinning process models are shown in Figure 25, Figure 26 and Figure 27, respectively.

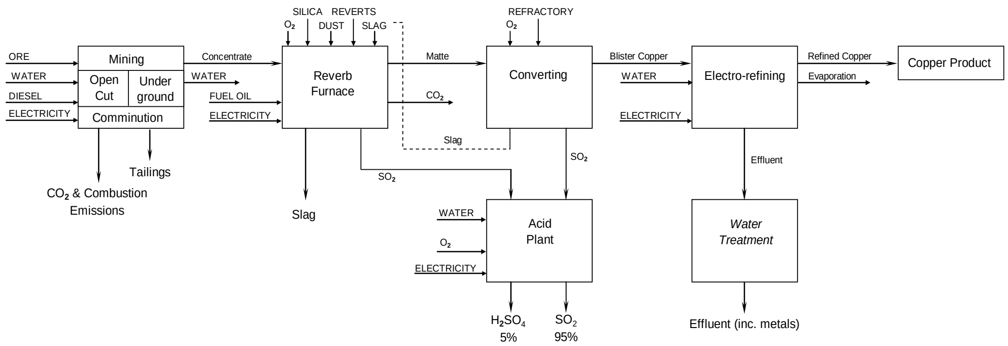

Figure 25: Process flowsheet for reverbaratory smelting[56]

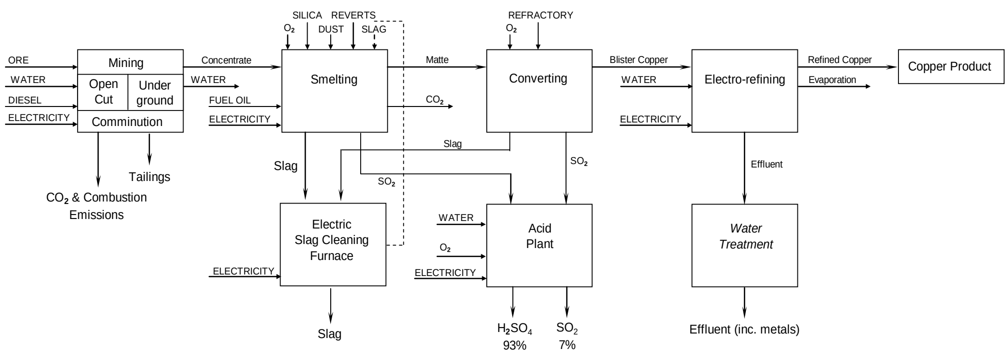

Figure 26: Process flowsheet for flash smelting[58]

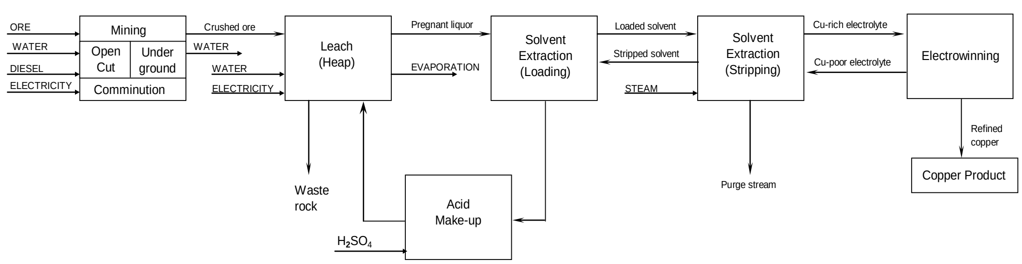

Figure 27: Process flowsheet for heap leach/solvent extraction/electrowinning[58]

The key assumptions underlying the above process flowsheets are shown in Table 10 below.

[56] Source: Giurco (2005)

Table 10: Key copper process modelling assumptions (Giurco, 2005)

	REVERB.	FLASH	HL-SX-EW
General			
Ore type	Sulphide	Sulphide	Sulphide
Ore grade	0.70%	0.70%	0.70%
Mining method	Open Cut	Open Cut	Open Cut
Target Cu production rate (tpa)	145,000	145,000	145,000
Overall Cu recovery	88%	88%	59%
Environmental			
Electricity consumption (kWh/t Cu)	5900	5200	9,900
Water consumption (t/t Cu)	112	101	166
Fuel oil consumption (t/t Cu)	0.48	0.15	-
Diesel consumption (t/t Cu)	0.31	0.31	0.52
SO_2 capture for H_2SO_4 production	5%	93%	-
H_2SO_4 make-up (t/t Cu)	-	-	1.7
Economic			
By-product value created			
Au recovery	70%	70%	0%
Ag recovery	70%	70%	0%
Average metal prices (Datastream, 2007)			
Cu (US$/t)	$ 7,123.56	$ 7,123.56	$ 7,123.56
Au (US$/oz)	$ 696.82	$ 696.82	-
Ag (US$/oz)	$ 13.38	$ 13.38	-

[Insert justification of the above assumptions as reasonable for this thesis]

4.2.2 Methodology

The eco-efficiency indicators for the above copper processing routes were computed using economic and environmental process performance information derived from preliminary material and energy balances. The procedure used is detailed in this section.

4.2.2.1 Economic performance assessment

The economic performance of the three copper processing routes described above as design alternatives was computed using revenue and cost information from various sources. Revenues generated were estimated using the annual metal production rate for each processing route (Cu, Au and Ag) and average 2007 metal trading prices from Datastream (2008). Operating and *annualised* capital cost estimates for each copper processing route were used as estimated by Biswas and Davenport (1994) for the year 1994. Values of the Marshall Cost index in 1994 and 2007 were used to account for cost inflation. In the initial analysis, the costs associated with income tax, interest, depreciation and amortisation were not taken into account. Equation 4 to Equation 5 below show the generalised formulae used to determine the economic benefit or value-add, B_i, from metal sales revenue, R_i, of m metals sold, less the capital costs, C_i, and operating costs, O_i, for each processing route, i, taking into account cost inflation between 1994 and 2007 using the Marshall and Swift index, γ.

$$B_i = R_i - \left(C_i + O_i\right) \cdot \left(\frac{\gamma_{2007}}{\gamma_{1994}}\right)$$

Equation 4

With

$$R_i = \sum_{k=1}^{m} F_{k,i} \cdot P_k = \sum_{k=1}^{m} \left(\phi_{k,i} \cdot \frac{F_{Cu,i}}{\phi_{Cu\_from\_ore,i}}\right) \cdot P_k$$

Equation 5

Where $\phi_{k,i}$ = Overall % recovery of metal *k* in processing route *i*

$F_{Cu,i}$ = Annual copper production rate (tonnes/annum)

P_k = Average trading price of metal *k* in 2007 (US $/tonne).

Capital costs were directly estimated for each unit operation as described above. Total operating costs were assumed to be composed of direct production costs, costs associated with metal sales and distribution to markets, management and overhead costs as well as project finance costs (primarily interest), as shown in Equation 6 below.

$$O_i = O_{production,i} + O_{sales,i} + O_{overheads,i} + O_{finance,i}$$

Equation 6

A complete set of assumptions and all collected raw data is available in Appendix B.1.

4.2.2.2 Environmental performance assessment

The environmental burden associated with each copper beneficiation process was determined for each of the four environmental impact categories investigated in this thesis. Initially, only *direct* environmental impacts were considered in this analysis, in line with a ‘gate-to-gate’ life cycle boundary. This was performed in order to understand the impacts that would seem to be of immediate importance to the decision maker *relative* to the overall impact of the process (which would necessarily incorporate indirect impacts). Indirect impacts are addressed in section 4.3.6. The methodological computation for direct impacts is outlined below.

a) Global warming potential

Direct greenhouse gas emissions associated with copper beneficiation processes arise principally from two key unit operations[57]: *mining* (due to the consumption of diesel) and *smelting* (from the combustion of fossil fuels – typically heavy fuel oil – as an energy source for the high-temperature furnaces) (Giurco, 2005). Average carbon dioxide emission factors and consumption rates for each process route have been used to estimate the amount of carbon dioxide emitted per annum from these processes, as shown in Equation 7 below.

$$F_{CO2,i} = F_{CO2,diesel} + F_{CO2,fuel\_oil}$$

$$F_{CO2,i} = \left(\frac{F_{Cu,i}}{\phi_{Cu\_from\_ore,i}}\right) \cdot \left[\left(\frac{M_{CO2}}{M_{diesel}}\right) \cdot \left(\frac{M_{diesel}}{M_{ore}}\right) + \left(\frac{M_{CO2}}{M_{fuel\_oil}}\right) \cdot \left(\frac{M_{fuel\_oil}}{M_{conc}}\right) \cdot \left(\frac{M_{conc}}{M_{ore}}\right)\right]$$

Equation 7

Where M_{CO2}/M_{diesel} = Average CO_2 emission factor from diesel combustion (kg CO_2/kg diesel)

$M_{CO2}/M_{fuel\ oil}$ = Average CO_2 emission factor from fuel oil combustion (kg CO_2/kg fuel oil)

M_{diesel}/M_{ore} = Average diesel consumption rate (tonne diesel/tonne ore)

$M_{fuel\ oil}/M_{conc}$ = Average fuel oil consumption rate (tonne fuel oil/tonne concentrate)

Further indirect greenhouse gas emissions are associated with the production of each energy source used in each process alternative, i.e. electricity, diesel and heavy fuel oil. In line with the constituency of the South African electricity mix, 95% of the electricity is assumed to be produced from hard coal, with an average emission factor of 1.01 kg CO_2/kWh (ESKOM, 2006). It has also been assumed that South African diesel and heavy fuel oil are produced from Middle Eastern crude oil, transported via high-sea tankers for processing at a coastal South African refinery (Frischknecht *et. al.*, 2007). Due to data limitations at this tactical phase of process design, pipeline and road transport emissions to the site have not been included in the analysis.

b) Dissipative water consumption

The open-pit mining, concentration and the electro-refining processes account for the bulk of aqueous waste streams leaving the mine, concentrator and smelting complex (Giurco, 2005). Since little of these waste streams is recycled back into the process, they can be regarded as 'dissipated' water streams. The total amount of water used by each processing route can thus be estimated using Equation 8 below.

[57] The milling stage is not included in this analysis since its associated greenhouse gas emissions arise from indirect sources (i.e. emissions from coal-fired electricity generation).

$$F_{H2O,i} = \left(\frac{F_{Cu,i}}{\phi_{Cu\_from\_ore,i}}\right)\cdot\left(\frac{M_{H2O,mining}}{M_{ore}} + \frac{M_{H2O,refining\_effluent}}{M_{ore}}\right)$$

Equation 8

c) Aquatic eco-toxicity

The definition of aquatic eco-toxicity used in this thesis is consistent with that offered by Goedkoop *et. al.* (2008), which incorporates the adverse environmental impact on freshwater aquatic ecosystems due to emissions into the air, water and soil. This metric therefore measures the eco-toxicity measure on *freshwater* sources, based on the assumption that effluent and atmospheric emissions from the mining operations considered eventually reports to freshwater sources such as rivers, dams and lakes. Terrestrial and marine (i.e. seawater) eco-toxicity as separately defined by Goedkoop *et. al.* (2008) are not considered in this analysis, given the relatively wide system boundaries and poor quality of information as argued by Basson and Petrie (2001) and elaborated on in Chapter 3 of this thesis. The cumulative eco-toxicity impact for each process route *i* was computed by aggregating the environmental eco-toxicity impact for each of the major and minor metallic elements *j* present in the ore, as shown in Equation 9 below. As described in Chapter 3, the reference eco-toxicity substance used was 1,4-dichlorobenzene. *Ecoinvent* normalisation data for each metallic element as developed by Frischknecht *et. al.* (2007) and revised by Goedkoop *et. al.* (2008) was used where available.

$$F_{DB,max} = \left[\sum_{j=1}^{l} \omega_{j,i}\cdot\left(\frac{F_{Cu,i}}{\phi_{Cu\_from\_ore,i}}\right)\right]\cdot\left(\frac{M_{14\_DB\_eq}}{M_{j,i}}\right)$$

Equation 9

Where $M_{14\_DB\_eq}/M_{j,i}$ represents the eco-toxicity normalisation factor for each metallic element *j* released in processing option *i* and $\omega_{j,i}$ is the mass concentration of each metallic element in the ore.

As Table 10 shows, the concentrations of these metallic elements in porphyry-type copper sulphidic ores were estimated from ranges offered by Broadhurst (2007a). The aquatic eco-toxicity normalisation factors and the concentration ranges for the key metallic elements present in the ore have been summarised in Table 11 below (further detailed information is available in Appendix A.1).

Table 11: Aquatic eco-toxity potentials and assumed trace element concentrations in plant tailings from porphyry-type copper sulphidic ores[58]

Component	Pyrometallurgical tailings		Hydrometallurgical tailings		Aquatic eco-toxicity
Major elements (mass %)	**Min**	**Max**	**Min**	**Max**	**kg 1.4-DB-equivalents**
Cu	0.5	1	0.5	1	1160
Fe	1	10	1	10	-
S	2	11	2	11	-
Moderately abundant elements (ppm)	**Min**	**Max**	**Min**	**Max**	
Zn	150	500	150	1600	7210
Pb	5	100	30	300	1110
As	2	550	5	1800	119000
Mo	4	450	15	1500	2620000
Bi	0.2	60	2	200	
Sb	0.2	60	2	200	1230
Cd	1	50	2	200	220000
Ni	1	50	8	150	2250000
Se	1	50	10	100	25300000
Trace valuable elements (ppm)	**Min**	**Max**	**Min**	**Max**	
Au	1	70	1	70	-
Ag	0.4	4	0.4	4	-

For the purpose of this analysis, the maximum metallic concentrations in the tailings were used to estimate the highest possible eco-toxicity impact from each processing route.

d) Resource depletion

The cumulative resource depletion impact for each process route *i* was computed by aggregating the resource depletion impact for each of the major and minor metallic elements *j* present in the ore, as shown in Equation 10 below. Antimony (Sb) was used as the reference substance, also based on *ecoinvent* normalisation data for each metallic element as developed by Frischknecht *et. al.* (2007) where available.

$$F_{Sb,max} = \left[\sum_{j=1}^{I} \omega_{j,i} \cdot \left(\frac{F_{Cu,i}}{\phi_{Cu\_from\_ore,i}} \right) \right] \cdot \left(\frac{M_{Sb\_eq}}{M_{j,i}} \right)$$

Equation 10

Where $M_{Sb\_eq}/M_{j,i}$ represents the resource depletion normalisation factor for each metallic element *j* that is mined and beneficiated using processing option *i*

[58] Source: Goedkoop *et .al.* (2008) and Broadhurst (2007a).

The resource depletion impact of each process alternative was also estimated using the same metallic elements presented in Table 11 above, based on *feed ore* concentrations as opposed to *tailings* concentrations as used in Table 11. This definition of resource depletion is consistent with that offered by Goedkoop *et. al.* (2008), on which the impact assessment data is based. In addition to direct resource depletion impacts, resource depletion associated with crude oil for the production of diesel and heavy fuel oil as fossil fuels used in the process is also included in this analysis. The inclusion of these impacts is verified in section 4.3.6. The resource depletion normalisation factors and these ROM ore concentrations for the key metallic elements present in the ore have been summarised in Table 12 below (further detailed information is available in Appendix C).

Table 12: Resource depletion potentials and assumed trace element concentrations in poryphry-type copper sulphidic ores[59]

Component	**Ore concentrations**		**Resource Depletion**
Major elements (mass %)	**Min**	**Max**	**kg Sb-equivalents**
Cu	0.5	1	0.000022
Fe	1	10	8.43E-08
S	2	11	
Moderately abundant elements (ppm)	**Min**	**Max**	**kg Sb-equivalents**
Zn	150	1600	0.000992
Pb	30	300	0.0135
As	5	1800	0.00917
Mo	15	1500	0.0317
Bi	2	200	0.0731
Sb	2	200	0.033
Cd	2	200	0.33
Ni	8	150	0.00018
Se	10	100	0.475

Having separately computed the economic (B) and environmental performance (E) for each process route, eco-efficiency indicators were then directly computed as a simple ratio between these performance criteria for each process route (i.e. $\Psi_i = B_i/E_i$). For this case study, the accuracy of the economic and environmental performance data was assumed to be at **order-of-magnitude** and **study estimate** levels, respectively (i.e. 40% and 25%, respectively). These indicators were then compared to the relative or normalised graphical representation of economic and environmental performance. These representations were compared in turn for distinguishability using the procedure outlined in Chapter 3 and in Appendix C (please refer to Appendix E for detailed process models and calculations underpinning this methodology section). The results of this analysis are presented below.

[59] Source: Goedkoop *et .al.* (2008) and Broadhurst (2007a).

4.3 Case study results

4.3.1 Economic performance assessment

Based on the above set of assumptions, the economic performance of each of the three considered copper beneficiation process flowsheets is summarised in Table 13 below to the appropriate number of significant figures. Further detailed information is available in Appendix B.1.

Table 13: Economic performance assessment for copper beneficiation process alternatives

	Reverb. Smelting	Flash Smelting	HL-SX-EW
COSTS			
Total major equipment costs (US$/annual tonne Cu)	$ 8,200	$ 7,500	$ 3,300
Working capital (US$/annual tonne Cu)	$ 800	$ 750	$ 330
Total initial capital investment (US$/annual tonne Cu)	$ 9,000	$ 8,300	$ 3,700
Total operating costs (US$/annual tonne Cu)	$ 2,700	$ 2,700	$ 2,200
REVENUE			
Cu sales (US$/tonne Cu)	$ 7,120	$ 7,120	$ 7,120
Ag credits (US$/tonne Cu)	$ 2,290	$ 2,160	-
Au credits (US$/tonne Cu)	$ 3,000	$ 2,800	-
Total revenue (US$/tonne Cu)	$ 12,400	$ 12,100	$ 7,120
VALUE-ADD (US$/tonne Cu) *Value Add = Total revenue – Total initial capital investment – Total operating costs*	**$ 700 ± 270**	**$ 1,100 ± 450**	**$ 1,220 ± 490**

The economic value-add for each process alternative has therefore been expressed as *profit* (i.e. revenue less costs) on an annual copper tonne basis. As the oldest and most capital-intensive technology currently being phased out of most modern operations, it is not surprising that the reverbaratory smelting process is the least economically attractive of all three options. The HL-SX-EW process costs much less than either of the pyrometallurgical processes; however, heap leaching yields less efficient extraction and recovery of copper from sulphide ores than does smelting (see Table 10). Furthermore, any precious metals present (e.g. silver or gold) in the ore are typically not recovered during solvent extraction and copper electrowinning[60] (Jergensen, 1999), whereas such additional value creation is possible in pyrometallurgical processes. Indeed, given the capital-intensive nature of smelting

[60] The number of technologies that have been developed to specifically address this are increasing, however (e.g. Peacey *et. al.*, 2003; Ally *et. al.*, 2001).

operations, gold, silver and molybdenum credits are often key financial viability drivers for copper beneficiation projects (Biswas and Davenport, 1994). This dependency can also be noted in Table 13 above: 43% and 41% of revenues can be attributed to these credits for the reverbaratory and flash smelting processes, respectively. Based on the above analysis, HL-SX-EW thus has a *cost* advantage over the reverbaratory and flash smelting processes, which in turn have comparatively stronger *revenue* positions. While the decision to implement a pyrometallurgical or hydrometallurgical process is usually dominated by the ore mineralogy and grade as decision criteria (Biswas and Davenport, 1994), this observation may become an important decision making criterion in cases where either processing route is technically viable e.g. the hydrometallurgical route might be preferred if long-term cost escalations are a concern, while the pyrometallurgical route would be the superior option if high metal commodity prices are expected to be sustained.

4.3.2 Environmental performance assessment

The analysis above indicates how economic performance may influence the selection and development of copper beneficiation processes. In this section, the environmental dimension of such an influence is analysed (the data supporting the results included in this section is available in Appendix A.1). Figure 28 below depicts the environmental performance of the reverbaratory smelting, flash smelting and HL-SX-EW processes for the four principal environmental impacts of interest in this thesis as motivated in Chapter 3 and investigated in this case study.

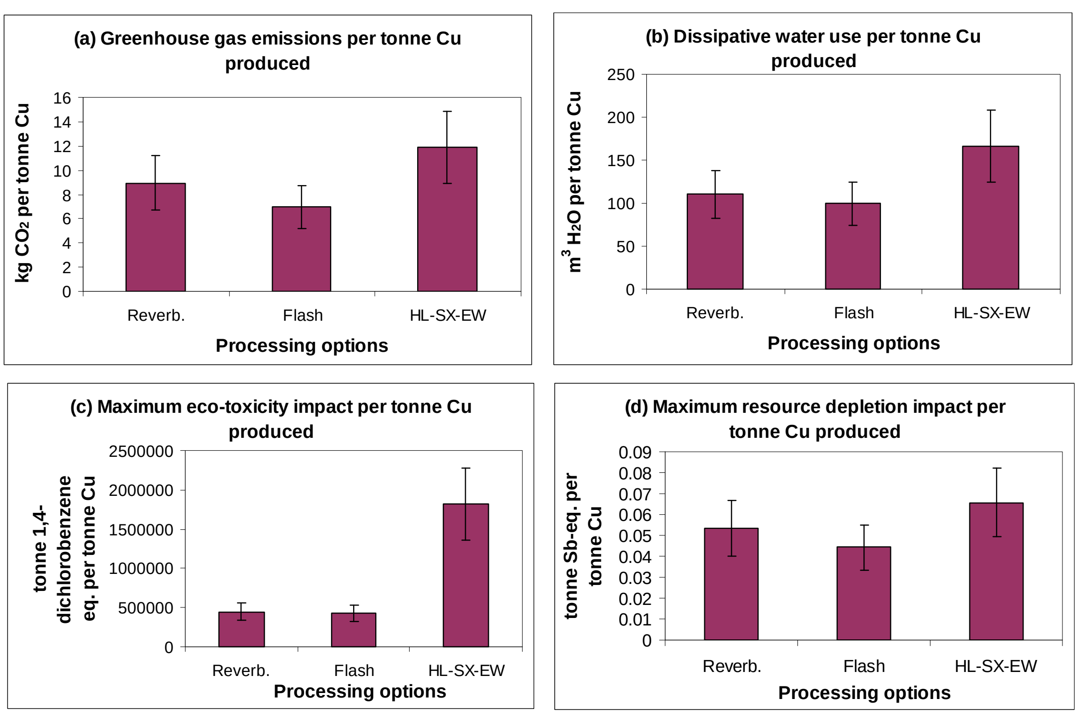

Figure 28: Environmental performance assessment for copper beneficiation alternatives

Importantly, both *direct* and *indirect* environmental impacts have been considered in the above figure. Figure 28 (a) above shows the global warming potential for each processing route in equivalent kilograms of carbon dioxide per tonne of refined copper. Figure 28 (b) communicates the *net* amount of water consumed by each process (i.e. total water use less the amount of water recovered) per tonne refined copper produced. The eco-toxicity impact for each process has been expressed based on kilograms 1,4-dicholorobenzene as a reference substance in Figure 28 (c) (as per common practice in environmental life cycle impact assessment). In this analysis, eco-toxicity arises from the mobilisation of toxic metals present in the ore as minor elements into aquatic streams. Due to the wide variability in the extent to which such toxic metals can dissolve into solution from the ore (Broadhurst, 2007a), the maximum eco-toxicity for each processing route has been considered in this analysis (i.e. assuming 100% dissolution). For similar reasons, the maximum resource depletion for each process alternative has been shown in Figure 28 (d), based on kilograms antimony equivalents as a reference substance. The process alternatives are then compared based on each environmental impact in turn below.

4.3.2.1 Greenhouse gas emissions

Direct emissions of carbon dioxide in copper beneficiation are attributed to the combustion of hydrocarbons in mining and smelting processes (Giurco, 2005). While diesel is used extensively in mining operations to power electric equipment and transport vehicles (and is therefore consumed in all three processing routes), reverbaratory smelting relies on fossil fuels such as heavy fuel oil to maintain the high temperatures required in the smelting furnace (Riekkola-Vanhanen, 1999). However, flash smelting harnesses energy from the combustion of sulphur in copper sulphide concentrates and consequently requires far less fossil fuel input, primarily used only for temperature control (Giurco, 2005). This accounts for the slightly higher CO_2 emissions noted for reverbaratory smelting when compared to flash smelting in Figure 28 (a) above. As evident from Table 10, the HL-SX-EW process consumes almost twice the amount of electricity used by pyrometallurgical processes due to the large amounts of electric current required in the electrowinning tankhouse. Combined with the lower copper recovery in HL-SX-EW (which results in higher ROM ore requirements to produce the same amount of refined copper as reverbaratory and flash smelting: 37% and 33% more ore is required for HL-SX-EW relative to reverbaratory and flash smelting, respectively), and despite the lack of direct hydrocarbon inputs into HL-SX-EW copper refining, CO_2 emissions for this process are much higher than those for reverbaratory and flash smelting, as evident from Figure 28 (a). This can be attributable to a far higher ROM ore requirement for HL-SX-EW to produce the same amount of copper metal as the reverbaratory and flash smelting processes (50% and 60% higher, respectively), translating to a net increase in the amount of diesel consumed during mining.

4.3.2.2 *Dissipative water consumption*

The decomposition of water in electrowinning (to yield oxygen at the anode) is the principal reason for the much higher water requirements of the HL-SX-EW process relative to the pyrometallurgical processing routes as evident from Figure 28 (b) (Ayres *et. al.*, 2002). Mining also consumes higher amounts of water for this process relative to smelting due to the increased ore requirements per unit of refined copper produced. Furthermore, the HL-SX-EW process is a net *consumer* of sulphuric acid since the amount of acid generated during electrowinning is typically lower than the amount required at the heap leaching stage, thus requiring an acid make-up stream which controls the leaching rate based on the acid concentration (and therefore water requirements). On the other hand, the pyrometallurgical processing routes are net *producers* of sulphuric acid, which can be sold as a by-product. While the acid plants in the smelting processes have significant water requirements, it is argued that this water is not used *dissipatively* since it becomes a component of an economically useful by-product (i.e. sulphuric acid) as it leaves the process. This water requirement is therefore not included in the performance metric above. This analysis therefore shows that the technical and economic viability of the HL-SX-EW process is much more dependent on the availability of quality process water than the pyrometallurgical processing routes. This places an even stronger imperative on the minimisation of dissipative water use for this process.

4.3.2.3 *Eco-toxicity*

Figure 28 (c) shows that HL-SX-EW has a far higher eco-toxicity impact relative to reverbaratory and flash smelting. This can be attributed to the low copper recovery for this process of 59%, implying that 41% of the copper present in the ore is returned to the environment in soluble and insoluble form in the waste rock. This is the key driver for acid mine drainage impacts often accompanying HL-SX-EW operations (Ayres *et. al.*, 2002). The low recovery of copper from sulphide ores has been the techno-economic driver for the use of bacteria in bioleaching processes to improve the copper extraction rate from the ore and limit the loss of copper to the environment (Brierley and Brierley, 1999). However, it must be emphasised that since this figure communicates the *maximum* eco-toxicity for each process, this ultimately represents only the highest potential for eco-toxicity damage. Environmentally sound heap/dump leaching management practices, such as those proposed by Brierley and Brierley (1999), may significantly lower this environmental risk.

While the above results indeed seem plausible, it is worth mentioning that the *ecoinvent* aquatic eco-toxicity normalisation scheme used in this analysis unfortunately does not have a normalisation factor for iron and sulphur, two major elements associated with copper sulphide ores. This is a critical shortcoming given that the deportment of iron and sulphur in a HL-SX-EW process is a key technical performance criterion while also significantly influencing the composition of the slag in pyrometallurgical processes (Ayres *et. al.*, 2002). However, it is well known that a much lower degree of control over the release of iron to the environment is typical of copper hydrometallurgical operations when compared to pyrometallurgical operations (Ayres *et. al.*, 2002; Riekkola-Vanhanen, 1999). It is therefore expected that the HL-SX-EW process would still have a higher aquatic eco-toxicity impact relative to the flash and reverbaratory smelting processes.

Nonetheless, the above shortcoming masks an even more important shortcoming: the inability of eco-efficiency to effectively prioritise which elements contributing to eco-toxicity impacts should be included in the environmental performance analysis. In this case study, the eco-toxicity impact assessment essentially assumes a prioritisation of elements to be included based on metallic concentrations in the ore, since only major elements and moderately-occurring elements are included in the study. Given the wide variability in the composition of ores treated across the world, the concentrations and morphology of metals present in the ore are expected to vary greatly. Furthermore, while a general correlation between the metal concentration and the eco-toxicity impact may exist, other rare elements that occur in extremely low concentrations within copper ores are notorious for their environmental hazards, e.g. selenium (Ayres *et. al.*, 2002). The contributions of these scarce elements to eco-toxicity impacts are currently not accounted for in this analysis, suggesting that another basis for prioritisation may be more useful. Hansen (2004) proposes that this impact assessment should be based on the *environmental risk potential* of a resource, which represents a more direct approach to assessing eco-toxicity. Broadhurst (2007a) suggests a generic approach to using the physical and chemical properties of the ore (of which metallic concentrations are but one criterion) to account for the *value potential*, *environmental hazard potential* as well as the *mobility potential* of metals within mineral-bearing ores. Such an approach may thus be more useful in better predicting the eco-toxicity impacts associated with mineral-bearing ores at the process design stage.

4.3.2.4 Resource depletion

The HL-SX-EW process also has the highest resource depletion impact of the three processing routes, as shown in Figure 28 (d). This is due to the relative inefficiency of the process in maximising copper recovery relative to reverbaratory and flash smelting (i.e. lower mineral resource efficiency). The higher diesel and fuel oil requirements in the *mining* section of the process (i.e. *not* in the metal extraction section) due to the comparatively higher ore ROM throughput for this process may also be a contributing factor. The shortcomings of eco-efficiency in predicting resource depletion impacts based on metallic concentrations as opposed to a more generic approach are also evident in this analysis. However, since it is likely that the concentrations of the metals in the ore have a much stronger influence on resource depletion than eco-toxicity (as the value potential of the metals would be a more important criterion in this category than the environmental hazard or the mobility potentials), this limitation is expected to be less of a critically decisive factor in quantifying resource depletion impacts as compared to eco-toxicity impacts.

4.3.3 Computation of eco-efficiency indicators

The economic and environmental performance of the reverbaratory smelting, flash smelting and HL-SX-EW processes analysed above is translated into eco-efficiency indicators in this section, thus constituting a basis on which the usefulness of such indicators relative to the absolute economic and environmental performance may be judged. These eco-efficiency indicators are shown in Figure 29 below.

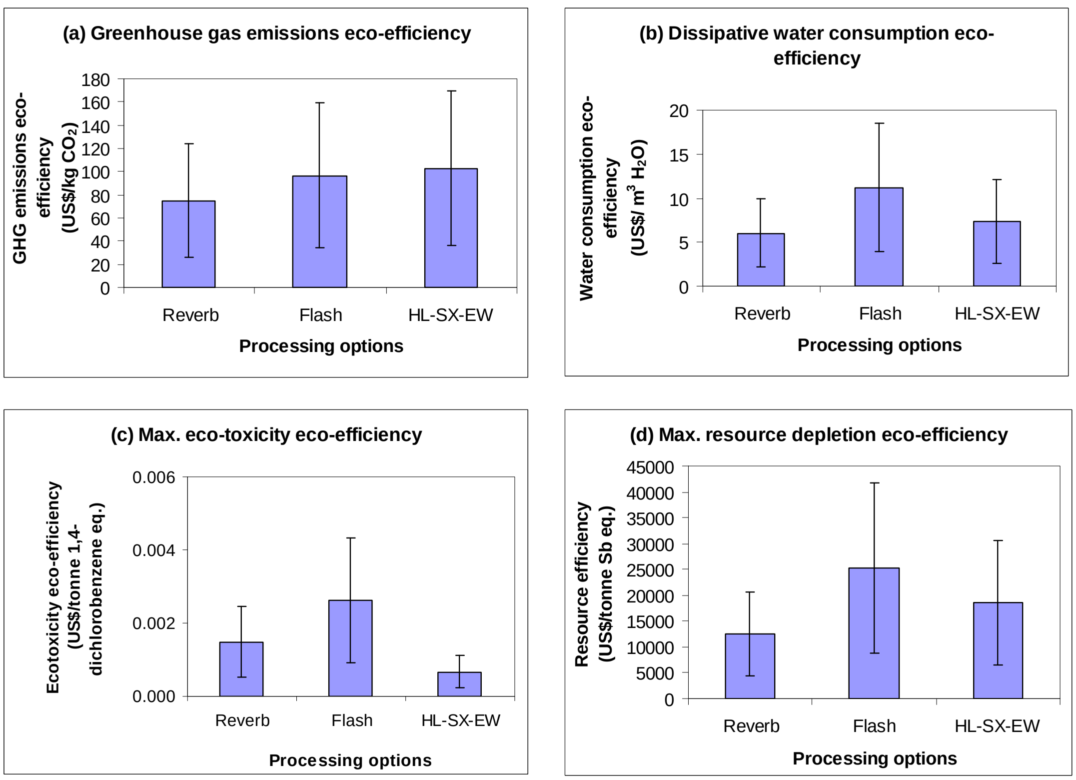

Figure 29**: Eco-efficiency indicators for copper beneficiation alternatives**

As can be seen above, eco-efficiency indicators combine economic and environmental performance information into a single performance metric for each environmental impact category, as value created per unit of environmental impact. Figure 29 shows that the flash smelting process has the highest eco-efficiency of the three process options in three of the four categories (HL-SX-EW is a competitive option to flash smelting when eco-efficiency in terms of greenhouse gas emissions is considered). This can be attributed to the combined effect of its strong economic performance (as shown in Table 13 above) with low environmental impacts in these impact categories relative to reverbaratory smelting and HL-SX-EW (in Figure 28 above). Reverbaratory smelting has the lowest eco-efficiency due its high environmental impact and its high-cost disadvantage when compared to flash smelting and the HL-SX-EW process.

However, of critical importance in this thesis is the need to *meaningfully elucidate* the eco-efficiency performance of various process alternatives for decision making. As evident from Figure 29 above, the usefulness of the eco-efficiency indicators is challenged by the high degree of uncertainty associated with these indicators. This arises from the fact that the economic data set in the tactical design decision context (particularly in the early phases) has an accuracy of ±40%, similar to that of study estimates (Basson and Petrie, 2001), while that of the material balances (from which the environmental performance profiles of the alternatives is computed) is ±25% (assumed to be similar to that of preliminary estimates in this case study). Therefore, in accord with uncertainty propagation theory, computing a ratio indicator from these data sets requires that the relative error ranges for each data set be added together. This therefore implies that the eco-efficiency indicators have an uncertainty range of approximately ±65%, which explains the large error bars noted in Figure 29 above. With such high uncertainty ranges, the ability to distinguish between the economic and environmental performance of the process alternatives above based on eco-efficiency indicators in this decision context is significantly hampered. The analysis indicates that the ratio nature of the eco-efficiency indicators serves to *compound uncertainty*, rather than reduce it. Given the relatively high uncertainty in the process data characterising this phase of the design procedure, this sensitivity to the quality of process data underpinning the performance analyses may be a critical shortcoming of eco-efficiency indicators. This assertion is further tested below, whereby the above (numeric) eco-efficiency indicators are compared to graphical approaches for representing the economic-environmental performance of the copper processing routes.

4.3.4 Graphical representations of eco-efficiency

Having represented the economic and environmental performance of the copper process route using numeric eco-efficiency indicators, it is now of interest to depict this performance in the graphical representation proposed in Chapter 3. The absolute eco-efficiency performance of each of the copper beneficiation processes is shown in Figure 30 below.

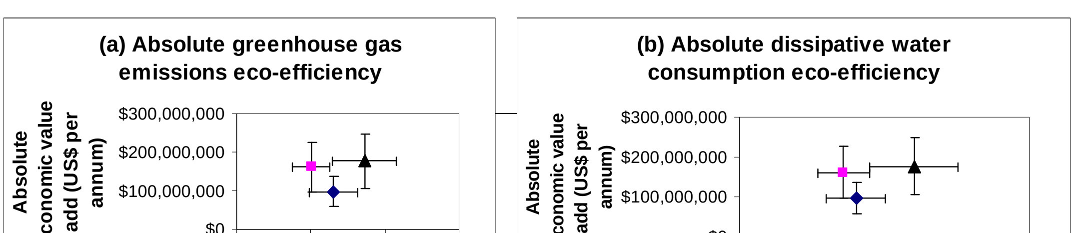

Figure 30: Absolute graphical eco-efficiency performance of copper process alternatives

The above performance depiction can also be shown on a normalised basis as proposed in Chapter 3. The relative graphical eco-efficiency of the copper processing routes is shown in Figure 31 below.

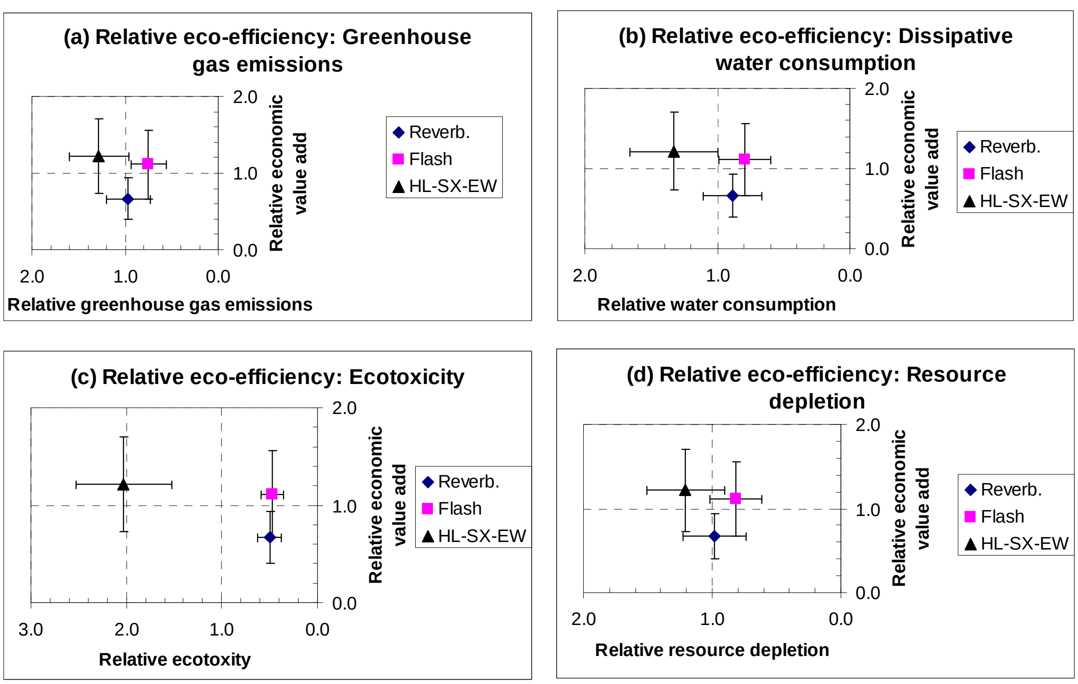

Figure 31: Relative graphical (normalised) eco-efficiency of copper process design alternatives

When Figure 30 and Figure 31 are compared, similar eco-efficiency performance characteristics can be observed: reverbaratory smelting offers the least economic return (within uncertainty) and HL-SX-EW is the least environmentally preferable processing route, with flash smelting appearing to offer the best compromise between economic and environmental performance. This similarity can be expected, given that Figure 31 is merely a homogenous normalisation of Figure 30. However, the ‘quadrant’ depiction embedded in the relative representation enables the decision maker to compare the performance of the design alternatives not only to each other (as can the absolute performance representation also), but also to a reference case or ‘weighted benchmark’ that is computed from the performance of *all* alternatives populating the decision space. In this manner, the decision maker can elucidate which alternatives have ‘good’ eco-efficiency performance (relative to the reference) taking into account the *entire* decision space, not just which alternatives have ‘better’ performance when compared to other alternatives. The relative graphical representation therefore provides an additional analytical tool to the decision space that can improve confidence in process selection (thereby improving the quality of decision making). As an example, in this case study, Figure 31 shows that despite the considerable uncertainty, flash smelting is the only processing route with an eco-efficiency performance that lies in quadrant II for all environmental impacts, further validating the claim made in the above sections as the preferable option. This advantage of relative graphical eco-efficiency depiction may be particularly useful and relevant when the technology options and choices are limited yet fundamentally different, thus making establishing a reference or ‘benchmark’ performance for absolute comparisons difficult (as opposed to different combinations of similar technologies with different flowsheet configurations, where a reference case may be easily defined). This is often the case during tactical process design as shown in this case study, as well as in early research stages of novel technology development (Basson and Petrie, 2001). Recent examples in the literature have also favoured this preference for normalisation (e.g. Michelsen *et. al.*, 2006; Saling *et. al.*, 2002).

Having discussed graphical eco-efficiency representations, it is now of interest to compare these with numeric eco-efficiency indicators as defined in this thesis. It has been mentioned above that despite the relatively large uncertainty bounds, the flash smelting process option has a high eco-efficiency with respect to all environmental impacts, since it can be found in quadrant II for all impact categories. This seems to be in agreement with observations made from eco-efficiency indicators computed in section 4.3.3 above if uncertainty is not taken into account. However, upon further inspection of Figure 31, it is evident that the distinguishability (i.e. the extent to which the performance ranges do *not* overlap) between the economic and environmental performance of the hydrometallurgical (i.e. HL-SX-EW) and pyrometallurgical processes (i.e. flash and reverbaratory smelting) has improved when compared to the eco-efficiency indicators shown in Figure 29. Moreover, it can be observed that the reverbaratory smelting process option is consistently inferior in economic performance to the flash smelting and the HL-SX-EW processes, since it never appears in quadrant II over its entire performance range for all environmental impact categories – an observation expected as discussed in section 4.3.1 above.

Critically, however, due to the large uncertainty ranges of eco-efficiency indicators, the two latter observations above cannot be made when the performance assessment is carried out solely using eco-efficiency indicators in Figure 29 above, despite the seemingly overall agreeable economic and financial superiority of the flash smelting process evident from Figure 30 and Figure 31. Furthermore, eco-efficiency indicators are not helpful in a case where design alternatives compared have similar indicator ratios, albeit different *absolute* values of economic and environmental performance. This is particularly evident when the water use and resource depletion eco-efficiency indicators of the reverbaratory and HL-SX-EW processes as options inferior to flash smelting are compared: given that their eco-efficiency indicators are comparable (Figure 29 above), their absolute economic and environmental performance become the key decision criteria. Eco-efficiency indicators cannot provide any more information in this regard; however, the graphical representation in Figure 31 clearly shows the economic disadvantage of reverbaratory smelting in this comparison, and the environmental disadvantage of the HL-SX-EW process in these two environmental impact categories. This critique therefore confirms that the ratio nature of numeric eco-efficiency indicators can mask some key insights into the performance characteristics of process design alternatives. This might be due to the relative magnitudes of the economic and environmental performance values (in terms of how they translate to a numerical eco-efficiency ratio) for each design alternative or the uncertainty associated with these performance values themselves. Careful application of these eco-efficiency indicators, particularly in design decision situations where only poor-quality data is available, seems to be a necessity for meaningful decision making in these cases.

4.3.5 Distinguishability analysis

The above interrogation of eco-efficiency in terms of numerical and graphical depiction of economic and environmental performance can further be validated with a distinguishability analysis as described in Chapter 3 and Appendix C. The results of this analysis are shown in Figure 32 below as distinguishability indices after Basson (2004). The indices have been computed for the ratio (numeric) and only the relative graphical representation, as the preferable graphical depiction in this case. Supporting data has been included in Appendix C.2.

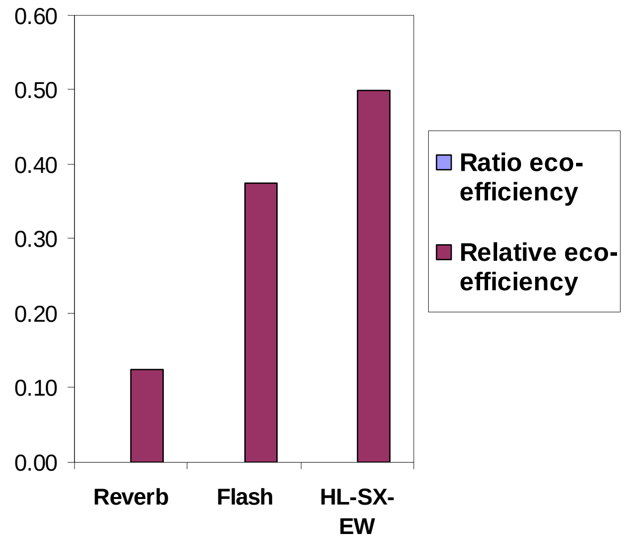

Figure 32: Ratio and relative eco-efficiency aggregated distinguishability indices in tactical design

The above figure shows that for all three process options, complete indistinguishability of the eco-efficiency performance of process design alternatives based on the numeric ratio indicators is observed, and the graphical (relative) representation of eco-efficiency has much higher distinguishability index scores relative to the numerical representation of eco-efficiency. This thus suggests that, in this case, more meaningful decision trade-offs can be explored with the latter rather than the former representation of the economic and environmental performance of the copper beneficiation processing routes investigated. However, given that an aggregated distinguishability index as close as possible to the value of unity is desired, the overall decision situation is still characterised by relatively weak distinguishability when Basson's convention is followed. This confirms the poor quality of the process information supporting decision making due to uncertainty. At this stage, a decision would therefore have to be made as to whether this level of distinguishability is acceptable and appropriate for the decision context at hand (Basson, 2004). An acceptable level of distinguishability would lead to a decision regarding which processing route should be further developed into a full flowsheet with more detailed material and energy balance information, therefore narrowing the design space into a single alternative (thus improving the resolution of the chosen processing route only) and moving the design procedure into the next phase. If this level of distinguishability is deemed unacceptable in this decision context, then another 'iteration' of this design phase would have to be performed, with efforts centred on improving the distinguishability of process alternatives (largely through, but not necessarily limited to, the reduction of uncertainty) till a decision on process selection can be made.

In this case study, the treated ore is assumed to be porphyry-type copper sulphide ore (a technical decision criterion favouring pyrometallurgical processing). A combination of *a priori* knowledge regarding the economic and environmental performance superiority of flash smelting over reverbaratory smelting (as described in section 4.1.2) and the preliminary eco-efficiency performance results presented above suggest that the flash smelting processing route may initially be construed to be the preferable option selected for further flowsheet development. However, the above analysis only refers to a comparative basis between the distinguishability of design alternatives using eco-efficiency indicators and using graphical representations of economic-environmental performance. Figure 32 shows that the aggregate distinguishability indices for *all* except one processing route have values less than the logical cut-off value of 0.5 defined in Chapter 3. Therefore, based on Basson's convention, the alternatives are only weakly distinguishable and the analysis indicates that it would not be worthwhile to continue with a comparative evaluation of the alternatives due to the level of uncertainty in the performance information. Further analyses would therefore have to be undertaken to reduce the uncertainty in the environmental and economic performance data before a decision on process selection could be confidently made.

4.3.6 Sensitivity analyses

Having compared eco-efficiency indicators to other more generic graphical approaches to representing the economic and environmental performance of process design alternatives, it is now useful to consider the usefulness of these performance metrics in the broader process design context. Drawing on the systems approach underlining the premises of this thesis, it is useful to recall from Chapter 2 that the *quality* and *quantity* of process design information as framed by the design basis were then key criteria of interest in process design towards environmental sustainability. Since the quality of design information as uncertainty has been directly discussed in this case study above, it is now of interest to consider whether eco-efficiency indicators can be meaningfully linked to well-known technical process design parameters that are contained in the design basis for a richer communication of technical, environmental and economic performance information, taking into account the manner in which the boundaries of the analysis are defined (i.e. the system boundary). This has been investigated through the use of sensitivity analyses, which are presented in this section. For the purposes of this analysis, only direct impacts are considered. Supporting data for this analysis is available in Appendix D.1.

4.3.6.1 Sensitivity of eco-efficiency indicators to technical design parameters

The ore grade of a mineral reserve remains one of the most important determinants of the viability of a minerals beneficiation project and forms a cornerstone technical criterion of the design basis for the evaluation of minerals development projects (Scott, 2002). The relationship between the economic and environmental performance values of the three copper processing routes and the ore grade has been investigated using a sensitivity analysis shown in Figure 33 below.

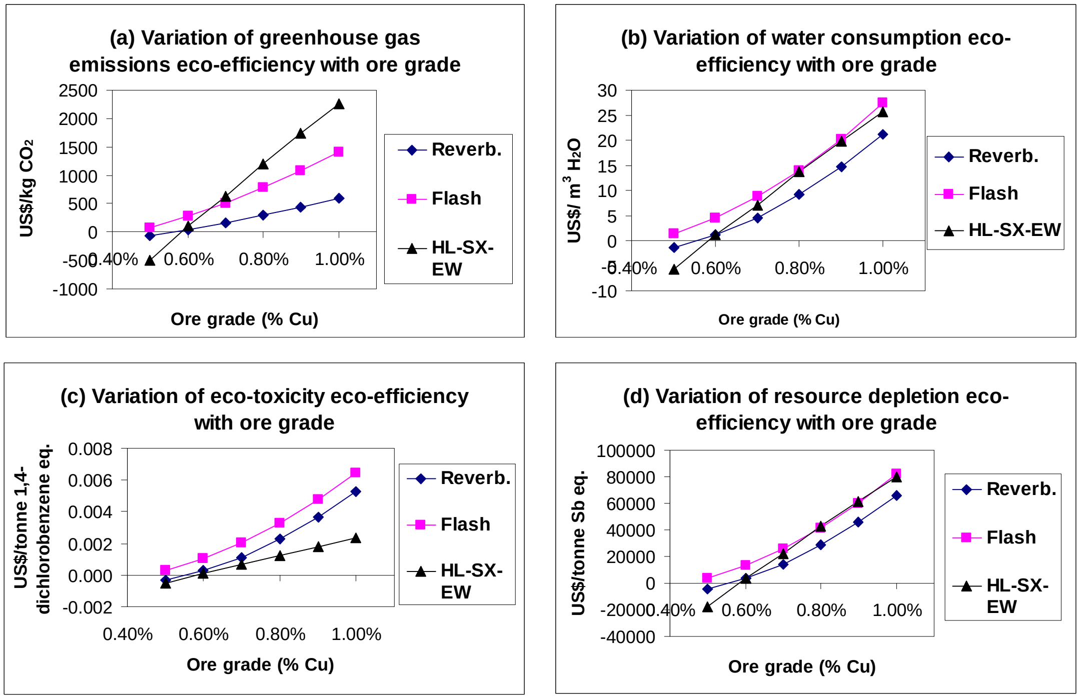

Figure 33: Sensitivity analysis of eco-efficiency indicators to the copper ore grade

Eco-efficiency indicators can also be computed as a function of the metal production rate (i.e. plant capacity). A sensitivity analysis of eco-efficiency indicators used in this case study to the copper production rate is shown in Figure 34 below.

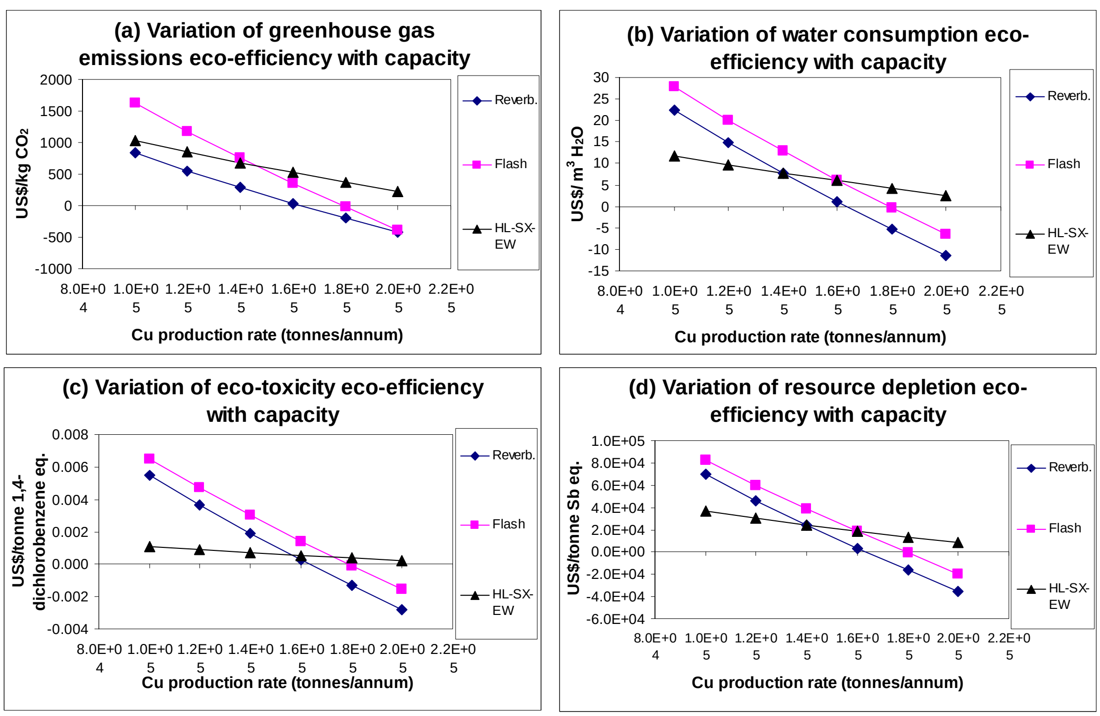

Figure 34: Sensitivity analysis of eco-efficiency indicators to plant capacity (copper production rate

As can be seen from Figure 33, *for a given copper production rate (i.e. fixed revenue)*, mining a mineral-rich ore results in an improved eco-efficiency compared to a relatively low-grade ore, principally due to lower input resource requirements which result in lower emission and effluent flowrates as well as reduced operating costs. However, if the metal production rate is increased as in Figure 34, the opposite effect on eco-efficiency is observed due to the combination of increased capital costs, operating costs and environmental impacts despite the increased revenue from metal sold. These sensitivity analyses demonstrate that eco-efficiency indicators can be used to represent and explore meaningful trade-offs between technical and economic-environmental performance information. For example, the sensitivity of eco-efficiency indicators to ore grade (Figure 33) shows that the economic performance of the process alternatives becomes marginal below a copper ore grade of 0.6%. The analysis also confirms that the flash smelting and the HL-SX-EW processes outperform the reverbaratory smelting process at almost all copper ore grade levels investigated. Various trade-offs also potentially exist between these two superior processes depending on the eco-efficiency category considered e.g. at 0.7% grade for the greenhouse gas emissions eco-efficiency. These trade-off points can also be observed in Figure 34 when plant capacity is the technical design parameter considered. The figure also shows that the eco-efficiency performance of the pyrometallurgical processes is more sensitive to changes in plant capacity than hydrometallurgical processes – an apt observation given the relatively capital-intensive nature of these processes and the strong influence of plant size on cost (Biswas and Davenport, 1994).

However, while the above discussion points to some value in using eco-efficiency indicators for tactical-stage minerals process design, previous critiques have highlighted that the quality of the information used (as manifest through uncertainty) also needs to be considered to ensure a meaningful assessment of eco-efficiency performance. Given the high degree of uncertainty associated with the eco-efficiency indicators used in this case study (not shown in the above figures for the sake of clarity), the confidence with which the above conclusions can be made is still debatable. However, it is argued here that this does *not* render eco-efficiency indicators to be of little use during tactical process design; rather, it is proposed that provided that adequate distinguishability in performance can be established, eco-efficiency indicators can better guide process selection by communicating a richer information set to decision makers as has been shown by the above sensitivity analyses. The quality of the information set and the decision making process as a whole would therefore be enhanced, particularly when used in conjunction with other decision analysis tools (e.g. multicriteria decision analysis tools).

4.3.6.2 Sensitivity of eco-efficiency indicators to system boundary definitions

The eco-efficiency analyses performed in sections 4.3.1 to 4.3.5 above have taken into account the environmental impacts associated with the energy requirements of the copper processing options considered in the form of electricity and fossil fuels (i.e. the production of diesel and heavy fuel oil), particularly when the global warming potential and the resource depletion impacts of each process are considered. The extension of the system boundary to include these impacts is therefore verified in this section. Figure 35 below compares the direct and indirect greenhouse gas emissions associated with each process design alternative.

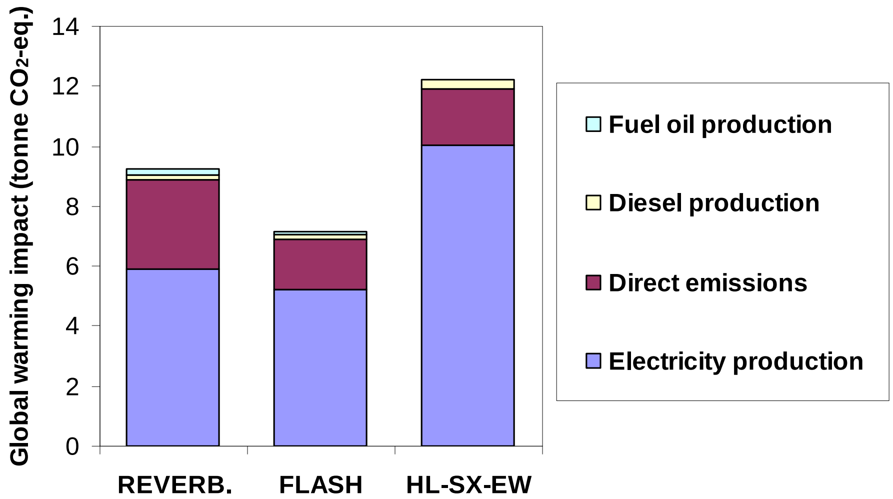

Figure 35: Comparison of direct and indirect greenhouse gas emissions of the copper process alternatives

It can be observed that for each processing route, the CO_2 emissions associated with coal-based power generation account for the largest share of all emissions that result from each process. This is an important observation given that these are ‘off-site impacts’ that are not under the immediate sphere of influence of the process design decision makers, but which nonetheless dominate the true global warming potential of the design alternatives considered. According to life-cycle thinking described in Chapter 2, a critical reason for including these seemingly external impacts is the need to avoid burden-shifting, i.e. shifting an environmental burden up or down the value chain (or life cycle). For example, if only direct emissions are considered in Figure 35, the HL-SX-EW process appears to have comparable direct emissions to the flash smelting option, even though its overall emissions are more than twice as large. However, despite the much-reduced milling requirements (and therefore reduced electricity consumption from mining processes) associated with HL-SX-EW (by approximately 70% according to Norgate and Rankin, 2000), electrowinning consumes about five times more electricity compared to electrorefining to produce the same amount of copper, and also uses a significant amount of energy for the mechanical agitation of solutions in solvent extraction (Norgate and Rankin, 2000; Biswas and Davenport, 1994). Considering direct emissions only would therefore merely ‘shift’ the bulk of the greenhouse gas emissions upstream to power generation, not reflecting its true environmental performance. These findings are also consistent with recent calls for the application of more life-cycle based thinking in the design of metal production processes (Norgate *et. al.*, 2007; Norgate and Rankin, 2004).

The direct and indirect impacts arising from the depletion of copper ore and liquid fuels has been compared in Figure 36 below. The depletion of the ore resource accounts for the largest share of the impact , at 75%, 80% and 84% for the reverbaratory smelting, flash smelting and HL-SX-EW processes, respectively. While indirect impacts in this category are not as dominant as those from electricity production in the global warming impact category, the figure shows that the contributions to resource depletion (the resource depleted being crude oil) from the production of fuel oil and diesel are still considerable. Interestingly, in line with the computation of the maximum resource depletion impact, the definition of resource depletion used in this thesis after Frischknecht *et. al.* (2007) takes into account the total amount of metal extracted from the ground, regardless of whether the metal enters the manufacturing value chain (i.e. is successfully sold as metal product) or deports back to the biosphere as toxic metal waste. This therefore translates to a maximum resource depletion impact that can only be an overestimation of the true impact within the re-defined system boundary. The significance of these indirect impacts despite this likelihood of overestimation therefore further strengthens the case for their inclusion in environmental performance analyses during minerals process design, as performed in this case study.

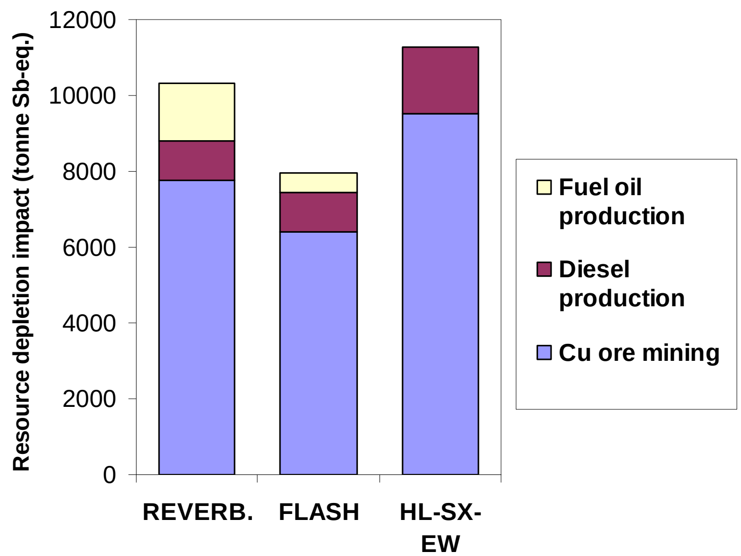

Figure 36: Contributions to resource depletion impacts from copper process alternatives

Given that each process design alternative as an anthropogenic activity is associated with other adverse environmental impacts that are often experienced beyond the site or plant's immediate boundaries (Stewart, 1999), the need to consider indirect environmental impacts that occur further up or down a given value chain has formed the for basis the uptake of the so-called 'life-cycle thinking' in process design, particularly within resource-based industries (e.g. Basson, 2004; Ayres *et. al.*, 2002; Notten, 2002; Stewart, 1999). The inclusion of these indirect impacts is also consistent with life cycle thinking, promoting an analytical paradigm shift from ‘gate-to-gate’ to more ‘cradle-to-gate’ analyses during minerals process design, as argued by Stewart (1999) and briefly described in Chapter 2. Life-cycle thinking is therefore a key cornerstone of the systems thinking approach that is the core argument for this thesis, emphasising the need to use process systems engineering concepts (such as uncertainty and boundary analyses) more proactively in better understanding the environmental performance of the minerals beneficiation processes to be designed and operated.

4.4 Summary

Chapter 4 has presented detailed findings from interrogating the application of eco-efficiency in a tactical design decision context during minerals process design, based on an economic and environmental performance assessment of copper beneficiation processes for preliminary process selection. The case study results have shown that while eco-efficiency indicators seem capable of *numerically* expressing the environmental and economic performance of the copper processing routes as design alternatives, the *meaningful* communication of these performance criteria for decision making is hindered by the ratio nature of the indicators, in both masking the *absolute* economic and environmental performance of the design alternatives and in compounding the uncertainty associated with these performance values. While graphical representations of eco-efficiency seem more capable of mitigating these challenges to support decision making that advances the design procedure relative to numerical eco-efficiency indicators, the distinguishability analysis shows that both graphical and numeric representations yield poorly distinguishable performance values for the processing routes, necessitating further reduction in uncertainty before a processing route can be selected for further flowsheet development. Other limitations that were noted with the use of eco-efficiency indicators include its lack of a detailed enough yet generic approach to better prioritise environmental impacts to be considered in an eco-efficiency analysis.

Nonetheless, in cases where a high quality of process data is available, the case study has also highlighted the value in using eco-efficiency indicators to meaningfully combine technical, environmental and economic performance information for useful interpretation by decision makers, thereby providing a good platform for an integrated approach that incorporates the use of multicriteria decision analysis techniques for better-informed and more transparent decision making. Eco-efficiency indicators were also shown as sensitive to the manner in which boundaries for the performance analyses were defined, further driving the need for systems thinking during environmental performance analyses.

Having explored the usefulness of eco-efficiency indicators in a tactical design decision context for primary metals production, eco-efficiency indicators in an operational design decision context are analysed next, in Chapter 5.

CHAPTER 5

Case Study 2: Eco-efficiency and the Operational Design Decision Context

This chapter explores the usefulness of eco-efficiency in an operational design decision context. A focused approach is employed, where eco-efficiency indicators are developed to quantify only **dissipative water consumption** as an adverse environmental impact in a gold beneficiation facility. Tailings dewatering circuits are generated with which water recovery from the tailings can be maximised. These circuits are therefore the key process design alternatives whose economic and environmental performance (the latter being relative to dissipative water consumption only) are analysed.

5.1 Background and description of the case study

5.1.1 Background to tailings dewatering process technologies

With the recent acknowledgement of the fast depletion rate of global natural water resources (UNESCO, 2003) and the importance of water in the minerals beneficiation sector (IIED, 2001), the efficient use of water within these processes has come to the research forefront. The dewatering of tailings from minerals beneficiation solid waste streams prior to their disposal in tailings dams forms a critical step in the recovery of water within a metallurgical process, often pervasively influencing the water balance over the entire mining operation (Mwakyusa, 2007). Furthermore, the recent spate of spectacular structural failures of tailings dams, such as those documented by Boger and Hart (2008), have added another dimension to the sustainability challenges facing mining companies, often resulting in unprecedented environmental damage, loss of lives and livelihoods to surrounding communities and a consequent loss of the 'social license to operate' for mining houses. This has consequently formed robust research, corporate strategy and operational management agendas for improving techno-environmental performance from tailings dewatering circuits nested in these operations, in which water is recovered and recycled. These agendas set the overall context for the research carried out as part of this case study.

Gravity thickening has long been a cornerstone of tailings dewatering processes (Mwale *et. al.*, 2005). Thickening within the minerals industries entails the removal of process water from a solid waste stream in a metallurgical operation through sedimentation and/or clarification, disposing the thickened tailings through the underflow and recycling the water back to the process plant for re-use (Azam, 2004). In addition to saving water, tailings are also thickened to maximise the solids content of the tailings storage facility to ensure the stability of the dam, typically measured through the yield stress or yield strength (Azam, 2004). Thickened tailings generally can be disposed of in tailings dams in three forms: a *slurry* (the most typical and oldest form of tailings disposal), a *paste* or as a much thicker *cake*. Figure 37 below qualitatively shows how the solids concentration of a typical tailings stream is related to the yield strength of the material.

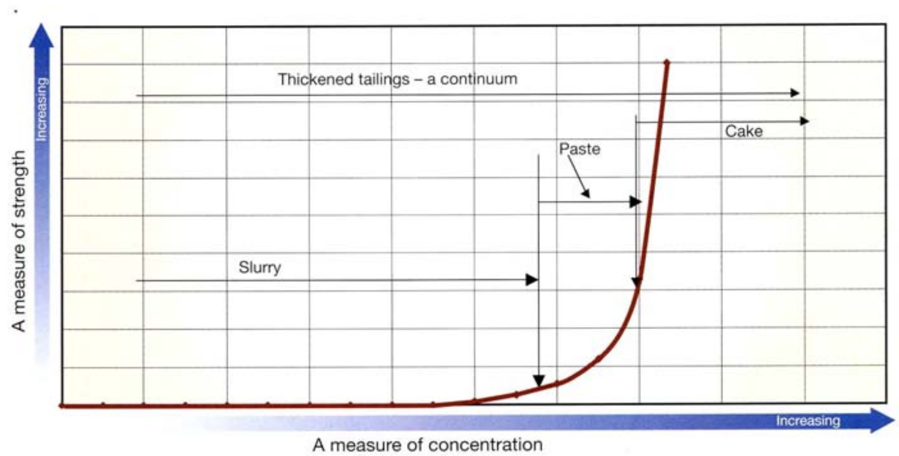

Figure 37: A typical yield strength concentration curve for a tailings suspension[61]

Technological developments in gravity thickening have been directed at designing thickeners that can produce higher solids concentrations in the underflow (i.e. high underflow densities). An alternative approach has been the use of other dewatering techniques such as filtration, which produces a cake, in conjunction with a thickener (Meggyes, 2004). Within the improved gravity thickening approach, an important technique has been the addition of *flocculants* to tailings. These are high molecular weight polymers that aid in enhancing settling rates of suspended mineral solids through particle agglomeration in thickeners (Bedell *et. al.*, 2002). In addition to achieving a higher underflow concentration over the same time period than a thickener not using flocculants, with an increased setting rate in a thickener, a smaller settling area is required, decreasing the capital costs of the thickener. While flocculants are typically added to most modern conventional thickeners, a more intensive utilisation of flocculants has, amongst other technological advancements, created a new suite of thickening technologies, such as *high rate*, *ultra-high rate* and *deep-bed* thickeners, whose settling zones are far deeper and narrower than those of conventional thickeners. This technological evolution of thickener design is shown in Figure 38 below.

[61] Source: Jewell *et. al.* (2006)

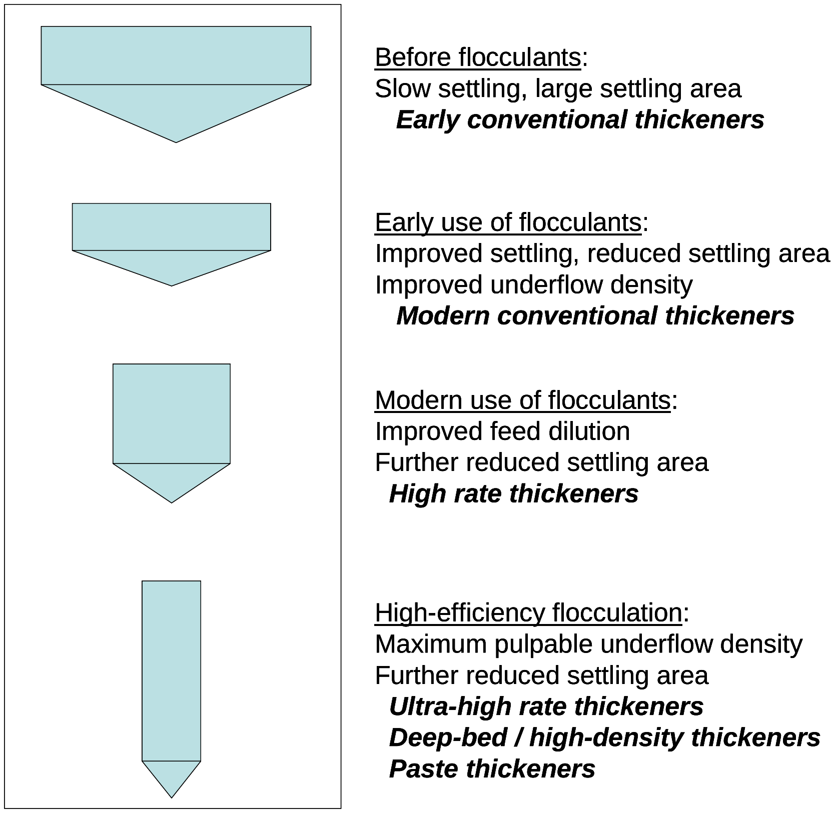

Figure 38: The technological evolution of thickener design[62]

With these developments, combined with the water management imperatives outlined above, a key question that process engineers in already-existing operations have been faced with has been whether to 'retrofit-design' currently existing conventional thickener circuits with other dewatering techniques such as filtration, thus achieving *incremental* performance improvement, or to replace the entire or a part of the existing thickener circuit with these newer and more efficient thickening technologies, which leads to fundamental technological *step-change* in the performance of the circuit. These incremental vs. step-change paradigms to system performance underpin the decision question that technical management professionals at an existing gold beneficiation operation in Tanzania were concerned with, which is used as a case study for interrogating eco-efficiency in this thesis. This operation, the Golden Pride Project, is described briefly below.

5.1.2 The Golden Pride Project, Tanzania

In addressing the water-related sustainability challenges in tailings dewatering described above, Mwakyusa (2007) compared the techno-economic and environmental performance of *wet tailings* and *filter cake tailings* disposal techniques, using the Golden Pride Project operation in Tanzania as a case study. This thesis seeks to refine this work by developing

[62] Source: Bedell *et. al.* (2002)

more meaningful process alternatives and re-assessing their economic and environmental performance more rigorously in terms of eco-efficiency. A brief description of the Golden Pride project is provided below.

The Golden Pride Project (GPP) is a gold processing and beneficiation operation wholly owned by Resolute Tanzania Ltd. It is located in the Tabora region of western Tanzania, approximately 750 km north-west of Dar-es-Salaam and 200 km south of Lake Victoria[63] (Resolute Gold Mining, 2007). In November 1998 it was the first modern mine to begin operation in Tanzania[64] (ICMM, 2006). During 2007, the operation produced 138,421 ounces of gold bullion from treating 2.51 million tonnes of ore with an average head grade of 1.94 g/t at a recovery rate of approximately 89% (Resolute Gold Mining, 2007).

The processing plant for the operation can be divided into three principal sub-processes according to Mwakyusa (2007):

i) *Ore preparation* – consisting of size reduction (crushing and milling) and classification (screening and centrifugal separation using hydrocyclones) processes
ii) *Gold extraction* – using a carbon-in-leach (CIL) circuit with seven tanks in series and cyanide as the leaching agent.
iii) *Gold recovery and purification* – employing an elution circuit and a pyrometallurgical (smelting) unit

A simplified version of the GPP plant flowsheet is shown in Figure 39 below:

[63] Source: http://www.resolute-ltd.com.au/op_goldenpride.html
[64] Source: http://www.icmm.com/casestudy.php?rcd=42

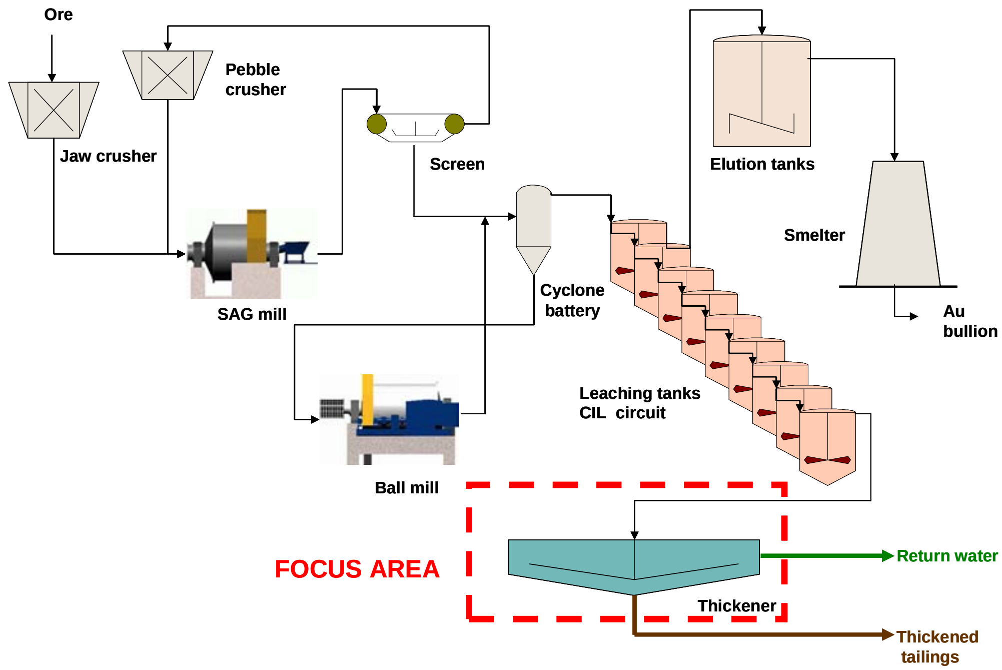

Figure 39: Golden Pride Project block flow diagram (with focus area)

The process model describing the focus area in the above flowsheet developed by Mwakyusa (2007) was extended in this thesis to interrogate eco-efficiency in an operational design decision context. Details on flowsheet development and the key assumptions made are provided in the following section.

5.2 Model development and principal assumptions

5.2.1 Flowsheet specification and principal assumptions

At the time of the initial study, the Golden Pride tailings dewatering circuit was comprised of two conventional thickeners that recovered water and cyanide from the tailings from the last tank in the CIL circuit before this material was sent to a wet tailings dam for disposal. To develop the dewatering circuit options as process alternatives, Mwakyusa (2007) used various combinations of four types of dewatering technologies, i.e. a *thickener*, a *lamella clarifier*, a *hydrocyclone* (thickened tailings or 'wet' disposal technologies) and a *filter* (a cake or 'dry' disposal technology) to generate six feasible dewatering flowsheet configurations[65].

For the purposes of this thesis, it is acknowledged that while Mwakyusa's initial case study was constructed based on an existing operation to be as realistic as possible, in reality the use of the lamella clarifier for dewatering as described in the case study would not be

[65] Flowsheet development and circuit selection methodology was based on a decision tree utilising the final moisture content (%) as the primary assessment criterion and EIA and NPV assessments as secondary criteria.

technically applicable for tailings dewatering since it is designed to handle dilute suspensions and therefore would be more suitable for clarification rather than thickening (Copeland, 2008[66]). Filtration is generally also considered an expensive option that is not widely used for dewatering tailings within the minerals industry (Copeland, 2008; Patterson, 2008[67]). On the other hand, the use of high rate thickeners over conventional thickeners for both new and existing operations has been increasing (Bekker, 2008). It is for these reasons, therefore, that the high rate thickener is used in this thesis instead of the lamella clarifier. As mentioned in section 5.1.1 above, high rate thickeners are designed for thickening and can achieve similar underflow solids concentrations as the lamella clarifier. Since environmental and economic performance (as compared to technical performance) are the central themes of this thesis, this modification of the case study is not expected to significantly influence the outcomes of this study in as much as these are meant to shed light on the usefulness of eco-efficiency indicators in minerals process design.

The revised six flowsheet permutations considered in this case study are shown diagrammatically in Figure 40 below:

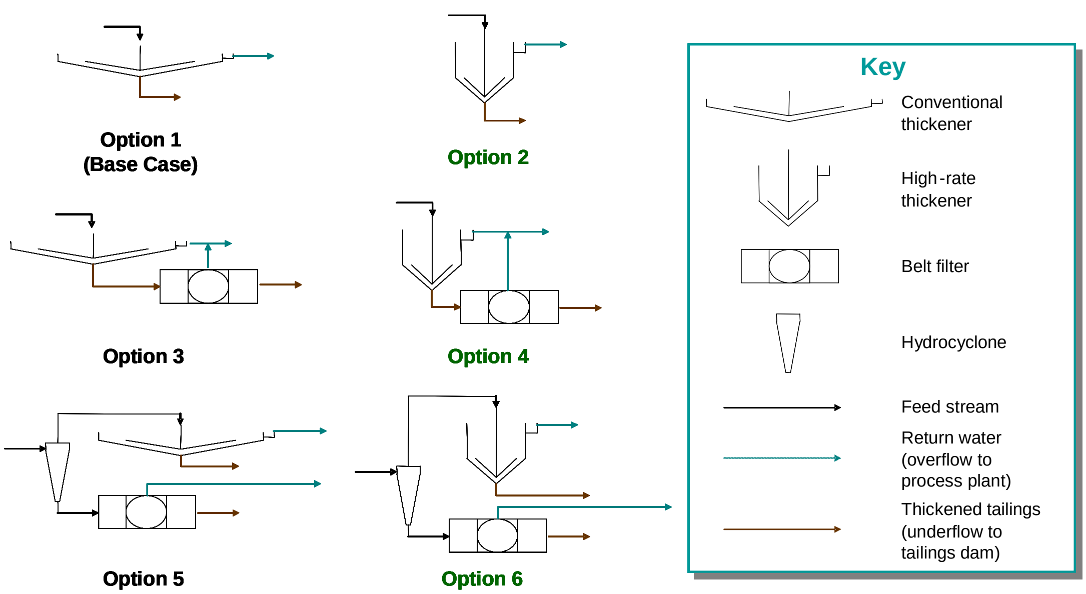

Figure 40: Feasible dewatering circuits investigated

In the initial assessment by Mwakyusa (2007), the simultaneous consideration of techno-economic and environmental performance for each flowsheet was achieved through a *net present value (NPV)* analysis (consisting of capital expenditure, operating costs and financial return in the form of water and cyanide savings through recovery and recycling) as the economic performance criterion, and *water loss* to tailings (expressed as the tailings moisture content in wt %) as the environmental performance criterion. It was conclusively

[66] Mr Andrew Copeland, Principal Engineer: Infrastructure & Tailings, Anglo American, 16/11/2008
[67] Dr Angus Paterson, Managing Director: Paterson & Cooke Consulting Engineers, 12/8/2008

demonstrated that filter cake tailings disposal techniques are environmentally superior to wet tailings disposal techniques since lower water loss was experienced with these flowsheets. However, a sensitivity analysis showed that the economic attractiveness of the proposed circuits employing filter cake disposal techniques was contingent on the water price or tariff. The simultaneous consideration for techno-economic and environmental performance is consistent with the eco-efficiency concept defined above and will therefore be used as a basis with which eco-efficiency indicators can be explicitly defined.

5.2.2 Methodology

The eco-efficiency indicators for the above dewatering circuit alternatives were determined based on two sets of criteria: **net present value (NPV)** for the economic *benefit* (numerator) and **(absolute) quantity of water lost**[68] from each dewatering circuit for the environmental *cost* (denominator), as outlined below. A complete set of assumptions and all collected raw data is available in Appendix A.2 and Appendix B.2. Detailed process models and calculations are available in electronic format in Appendix E.

5.2.2.1 Economic performance assessment

The Net Present Value was used to determine the economic benefit that would be derived from each dewatering circuit option as shown in Equation 11 below.

$$NPV_i = -C_i + \sum_{t=1}^{T} \frac{\left(R_{t,i} - O_{t,i}\right)}{(1-r)^t}$$

Equation 11

Where: C_i = total capital cost for the dewatering equipment in option *i*

$O_{t,i}$ = total operating costs for option *i* in year t

$R_{t,i}$ = avoided water raw material cost for process option *i* in year t

r = project discount rate

T = time horizon of the project

Equipment cost estimates were derived from empirical correlations and estimations as provided by Matche Consultants (2007), while operating costs were estimated from heuristics-based data from Metso Minerals (Sandgren *et. al.*, 2004) and STOWA Technologies (2006). Water savings achieved were based on a Tanzanian bulk water price of US$ 0.35 per ton

[68] The use of absolute water loss rather than a relative water loss expressed as wt % solids in the initial assessment is considered a more direct approach to determining the environmental impact associated with each dewatering circuit option, while also consistent with the classical definition of the eco-efficiency indicator offered by the WBCSD (2000).

(2006 value). The discount rate was set at 13%[69] and a project timeline of 8 years was assumed, based on the expected length of the remaining life of mine for the operation over which the new dewatering circuit would operate.

5.2.2.2 Net (dissipative) water loss

The amount of water loss to the environment for each circuit option was determined through an overall water balance around the circuit and the tailings dam. For this case study, the *total dissipative water loss from the dewatering circuit* is made up of evaporation and seepage, as detailed in the water balance model described in Appendix A.2 after Mwakyusa (2007). Table 14 below shows key mass balance assumptions for the dewatering equipment in the circuits.

Table 14: Summary of key information for the overall water balance

Key overall assumptions		**Value**
Water to tailings from rain (t/day)		Negligible
Daily specific evaporation rate (mm/day)		5.76
% Tailings water lost due to seepage		5
% average tailings water return from decanting		31
Equipment specifications	**Parameter**	**Value**
Hydrocyclone	Feed solids %	50
	Underflow solids %	60
Thickener	Underflow solids %	60
	Overflow solids %	0
High rate thickener	Underflow solids %	70
	Overflow solids %	0
Filter	Final cake moisture %	21
	Overflow solids %	0

The dissipative water consumption eco-efficiency indicators were computed as a ratio of the above two criteria (i.e. $\Psi_i = B_i/E_i$). For this case study, the accuracy of the economic and environmental performance data was assumed to be at **preliminary estimate** and **definitive estimate** levels, respectively (i.e. an accuracy of 12% and 6%, respectively). The techno-economic and environmental performance of the dewatering circuit options in Figure 40 above were compared based on a classification according to the type of the main or primary dewatering equipment (i.e. whether the option uses a conventional thickener or a high rate thickener). Therefore, two 'sets' of dewatering circuit options emerge – one including all options using a *conventional thickener* (called **set A** and including options 1, 3 and 5), and one including all options using a *high rate thickener* (**set B**, including options 2, 4 and 6). The results of this analysis are presented below.

[69] The discount rate was based on GPP management feasibility study practices, which assume a 20% discount rate, taking into account an inflation rate of 7% (Mwakyusa, 2007). However, it must be noted that this value is relatively high, since typical values for discount rates of around 6% above inflation are commonly assumed. However, given the recent rise in global rates of return (von Blottnitz, 2008), the overestimation of the discount rate may not be as severe. Furthermore, this fact is not expected to significantly alter the objectives of this analysis since the discount rate is commonly applied across all dewatering circuit alternatives, and to all NPV values, are expected to be negative.

5.3 Case study results

5.3.1 Economic performance assessment

The economic performance of the dewatering circuit options investigated as design alternatives in this case study is shown in Figure 41 below (supporting data is included in Appendix B.2)

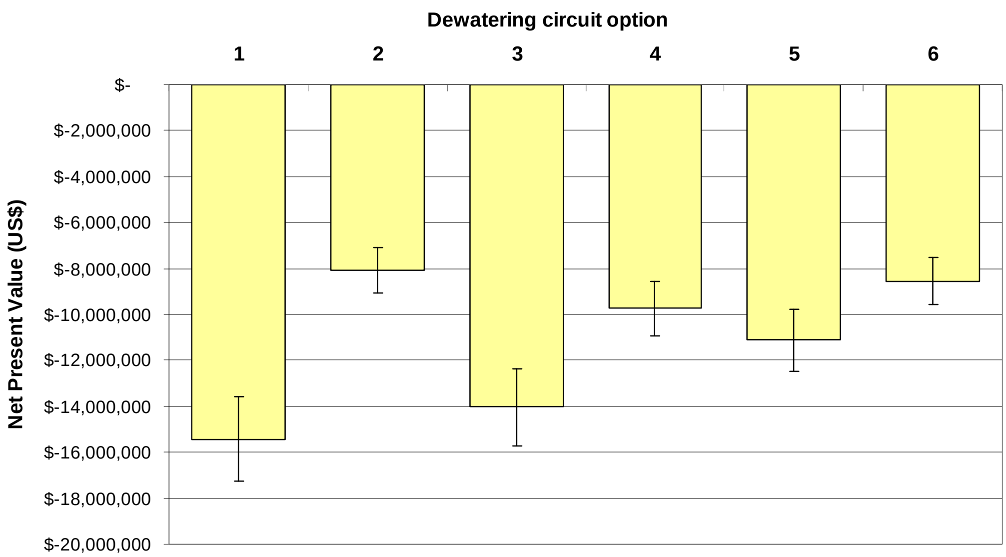

Figure 41: Economic performance assessment for gold tailings dewatering circuit alternatives

Since the water savings derived from each circuit will always be less than the capital and operating cost requirements at current water prices, all options show a negative net present value. When comparing options 1 and 2 above, high rate thickening yields a higher NPV than conventional thickening due to the lower capital cost requirement from the reduced area as well as a higher water recovery (and therefore increased water savings). This observation is in good agreement with the literature (e.g. Meggyes, 2004; Bedell *et. al.*, 2002). With the introduction of filtration in options 3 and 4, a significant cost reduction is achieved for the tailings dam, since cake disposal has lower capital costs and also requires far less maintenance (bench raising, draining, cleaning and re-piping etc.), which lowers operating costs (Wakeman, 2007; Metso Minerals, 2006; Mayer, 2000). Improved water savings are also achieved from avoided water loss. In option 5 and option 6, hydrocyclones were introduced principally to reduce the amount of slimes (defined as < 38 μm fine solids in this study) reporting to the belt filters, since these can result in severe clogging of the filter cloth and consequently inefficient cake washing (Mwakyusa, 2007). This is called 'partial classification', and is a well-established technique for increasing thickening and filtration rates in dewatering circuits (Brackenbusch, 1994). The introduction of the hydrocyclones also results in a lower throughput requirement for the thickening section of each circuit (since only the cyclone overflow reports to the thickener(s)), which in turn results in lower capital cost requirements.

In each of the equivalent circuits within the above sets of dewatering circuits (i.e. option 1 vs. option 2, option 3 vs. option 4 and option 5 vs. option 6), the economic performance superiority of the high rate thickener is consistently apparent. However, diminishing economic returns are experienced as more dewatering units are added onto the circuit flowsheet: the 'margin of improvement' is much greater between option 1 and option 2 than it is for option 5 and option 6. This trend is also noted *within* the conventional thickener circuit set i.e. by the notation used in this thesis, set A: $B_1 < B_3 < B_5$. This is not evident within the high rate thickener circuit set (set B), however. An interesting trade-off analysis would therefore have to be carried out to determine whether it would be worthwhile to switch technology to installing a high rate thickener or to augment the current conventional thickener circuit with filtration and hydrocyclone units. While this analysis suggests that installing a new high rate thickener would yield the highest economic return, the basis of the analysis warrants the need for caution, since other associated costs likely to arise have not been taken into account (e.g. conventional thickener decommissioning and disposal costs, installation costs, circuit reconfiguration costs and other hidden costs). A more practical solution might therefore be to reconfigure the current circuit to include a bank of hydrocyclones and belt filtration circuit (i.e. option 5).

5.3.2 Environmental performance assessment

The water loss performance data for the dewatering circuit alternatives are compared in Figure 42 below (please refer to Appendix A.1 for supporting data).

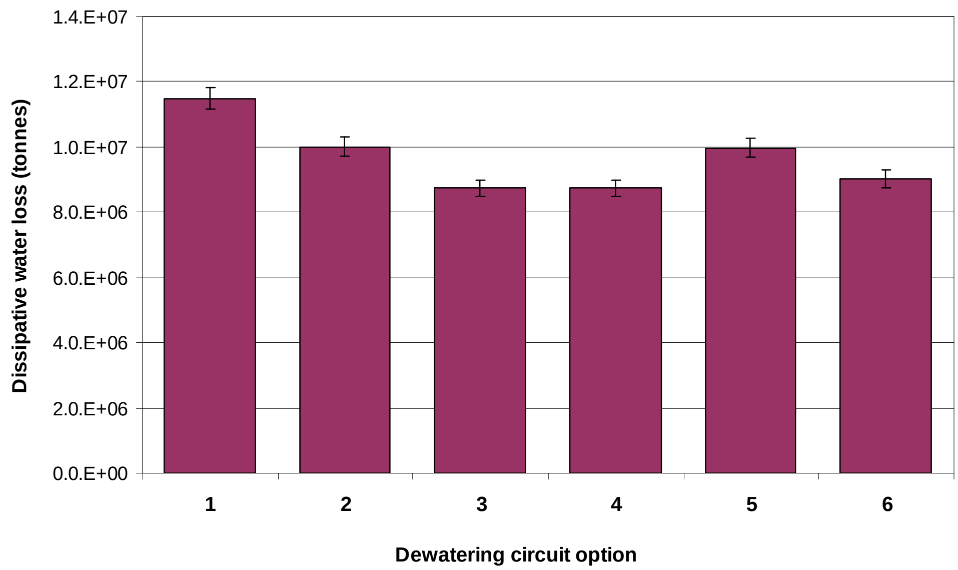

Figure 42: Dissipative water consumption for gold tailings dewatering circuit alternatives

It can be noted from the above figure that there is a significant reduction in water loss across all alternatives relative to option 1 as the current base case, thus validating the original case for reducing water loss in the current circuit and the choice of design alternatives considered. The implementation of options 3 and 4 as dewatering circuits results in the lowest water loss (approximately 24% reduction relative to option 1), since a much higher proportion of the circuit feed is dewatered using filtration instead of the (relatively less efficient) thickeners, resulting in thicker tailings than those from option 5 and option 6. Interestingly, in terms of water loss, a high rate thickener circuit is equivalent to a conventional thickening circuit with hydrocyclones and filters. While the latter circuit may seem preferable to the installation of a new high rate thickener based on the cost concerns mentioned in section 5.3.1 above, reconfiguring a dewatering circuit is associated with its own set of challenges. Particularly, belt filters have been notorious for operational problems in tailings dewatering, limiting their application for this use. Filter cloth clogging, the blocking of bulk materials handling equipment and various supply chain issues (e.g. purchasing spares) have been reported with the use of belt filters (Meggyes, 2004; Bedell *et. al.*, 2002). Combined with their relatively high capital and operating cost requirements (Meggyes, 2004), this affirms that the expected water savings from dewatering using filtration needs to be carefully considered against a cost-benefit analysis taking into account the local conditions of the dewatering problem.

5.3.3 Eco-efficiency indicators

In this section, the above economic and environmental performance of the dewatering circuit alternatives is compared by means of an eco-efficiency indicator performance assessment. The eco-efficiency indicators for each dewatering circuit alternative are shown in Figure 43.

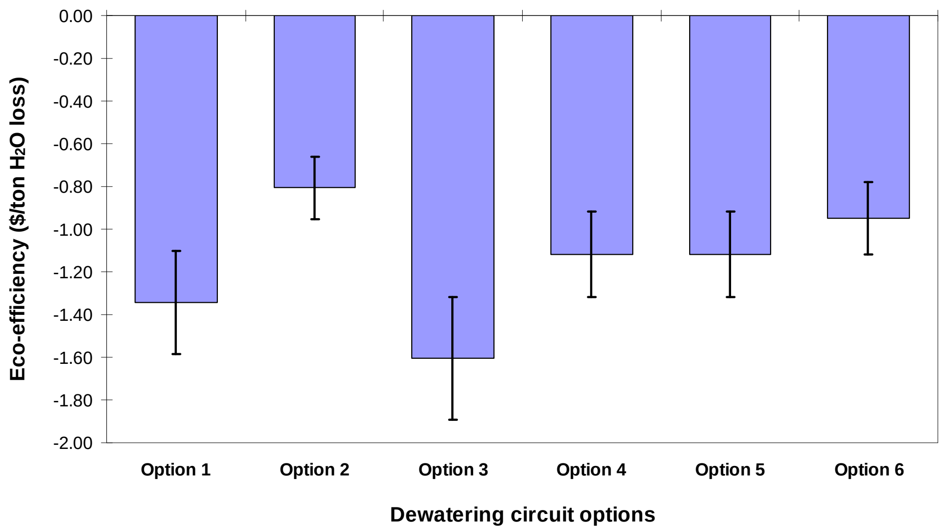

Figure 43: Eco-efficiency indicators for gold tailings dewatering circuit alternatives

Two key observations are evident from Figure 43. Firstly, dewatering circuit options 2, 4 and 6 (i.e. using the high rate thickener, or set B) have higher (or less negative) eco-efficiency indicators when compared to options 1, 3 and 5 (set A using the conventional thickener), which translates to a superior level of eco-efficiency. While this observation might affirm the technological superiority of the high rate thickener as discussed in the preceding sections, this might also merely be a manifestation of a similar trend noted in Figure 41 for economic performance, coupled with relatively small differences in water loss as can be seen in Figure 42. According to the above figure, option 5 and option 2 are the most eco-efficient flowsheets in set A and set B, respectively. This means that according to eco-efficiency indicators, installing a new high rate thickener circuit (i.e. without any other dewatering equipment) would be the most eco-efficient option if a technological step-change is desired, while augmenting the current conventional thickener circuit with a hydrocyclone bank *and* a filtration circuit would be the most eco-efficient option if incremental performance improvement is preferred.

Another observation pertains to the ability of eco-efficiency to inform decision-making *within* a technological frontier, i.e. choice of operating regime using the same technology or technology set. While a preference ranking for set B seems apparent (i.e. based on the notation used in this thesis, $\psi_2 > \psi_6 > \psi_4$), Figure 43 suggests that the option 3 has a lower eco-efficiency than option 1. This is a misleading observation since Figure 41 and Figure 42 demonstrate the techno-economic and environmental superiority of option 3 when compared to that of option 1 (option 3 results in far lesser water losses than option 1, and also looses significantly less money, making it an obvious preferred option). This observed shortcoming in the use of an eco-efficiency indicator is consistent with the reservations expressed by Michelsen *et. al.* (2006) regarding the use of numeric indicators such as these as sole decision analysis tools for considering the environmental performance of design alternatives.

Given the above potential drawback in the use of eco-efficiency indicators, it is therefore of interest to consider how graphical depictions of performance compare to numeric indicators as shown in this section. This comparison is performed next.

5.3.4 Graphical representations of eco-efficiency

The NPV and the cumulative water loss per annum (in tons) for the six flowsheet options that were analysed are plotted graphically in Figure 44.

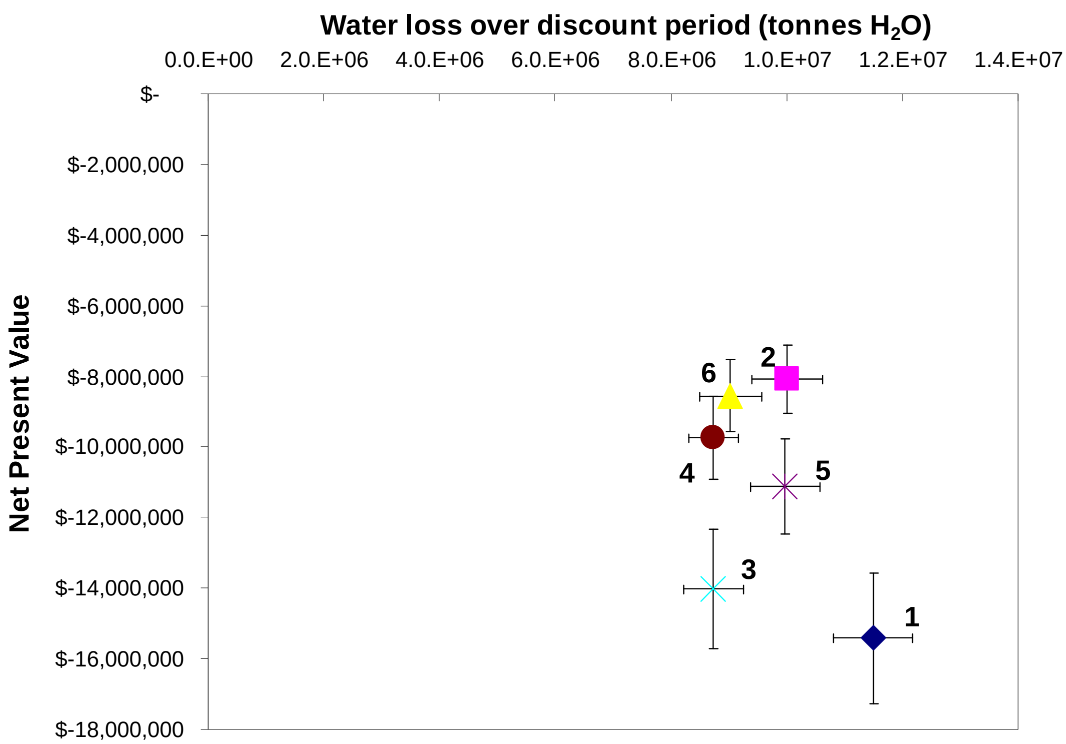

Figure 44: NPV vs. cumulative water loss over the life-of-mine from 2006

It can be observed that options 1, 3 and 5 are sub-optimal to options 2, 4 and 6. Importantly, option 1, the option using only conventional thickeners, is shown to be sub-optimal to all other options investigated in that it reports the highest water loss at the lowest economic benefit. Comparing option 1 and option 2 above, the technological superiority of the high rate thickener in recovering more water at a lower cost is immediately evident. The figure shows that replacing the conventional thickener in each flowsheet configuration (options 1, 3 and 5) with a high rate thickener would result in improvement on at least one of the two axes – a higher NPV and a lower water loss is achieved for flowsheet option 2 and option 6 (configurations equivalent to option 1 and option 4 using the conventional thickener), while option 4 improves the NPV from option 3, but at the same water loss. The figure therefore graphically depicts the role which technological innovation can play in improving both the environmental and economic performance of a base flowsheet: the replacement of the conventional thickener with a high rate thickener in the dewatering circuit results in more eco-efficient flowsheet configurations being achieved, translating the technological superiority of the high rate thickener from an individual unit to a process or systemic level.

Figure 44 above is therefore a translation of Figure 41 and Figure 42 onto a 2-dimensional decision space described in Chapter 3. According to Figure 44, option 6 has the highest eco-efficiency within set B (since it is the closest point to the origin on the design space by inspection), with option 4 and 2 having lower economic value return and higher environmental impact, respectively. The apparent superior performance of option 6 is evident when Figure 41 and Figure 42 are compared: option 2 has highest NPV (lowest *negative* NPV value) but also has the highest water loss of the three circuit alternatives employing high-rate thickeners, and option 4 has lowest NPV but also the lowest water loss. Option 6 has only a slightly higher water loss than option 4 but a much higher NPV (which is only marginally lower than that of option 2), giving it the highest eco-efficiency ratio overall.

However, the above explanation is *not* consistent with Figure 43, which suggests that option 2 has the highest eco-efficiency. Furthermore, the numeric eco-efficiency in Figure 43 suggests that option 4 has a similar eco-efficiency to option 5, while the graphical analysis in Figure 44 clearly shows that option 4 has a higher economic *and* environmental performance than option 5. By inspection of Figure 44, it can be noted that option 4 and option 5 lie on approximately the same line from the origin, consistent with the negative NPV case postulated in Chapter 3. This is particularly noteworthy given that these two options lie in different technology frontiers – option 5 belongs to the conventional thickener technology frontier (i.e. set A), while option 4 belongs to the high-rate thickener frontier (i.e. set B). In the preceding section, the comparison of the eco-efficiency performance of option 1 and option 3 has already shown that eco-efficiency indicators can be misleading even when applied within a technology frontier in the decision space. Therefore in this case, eco-efficiency has been shown to be potentially misleading in elucidating real performance information both *within* and *across* a technological frontier. This therefore highlights an important danger in misguidedly using generic eco-efficiency indicators as all-encompassing decision-making tools and suggests that a case can be made for the use of eco-efficiency with other valuation techniques to explicitly define the decision-maker’s preferences so that appropriate choices can be made. This might range from simple heuristics-based ‘rules of thumb’ within the industry to more sophisticated value measurement-based preference modelling techniques and value functions as described by Basson (2004). This is an important conclusion that is re-visited in Chapter 6.

Given that the above critique still lacks some clarity due to negative economic performance values, a normalised eco-efficiency analysis may therefore be useful as this would essentially overcome the problematic nature of negative eco-efficiency values. The results of this analysis are shown in Figure 45 below.

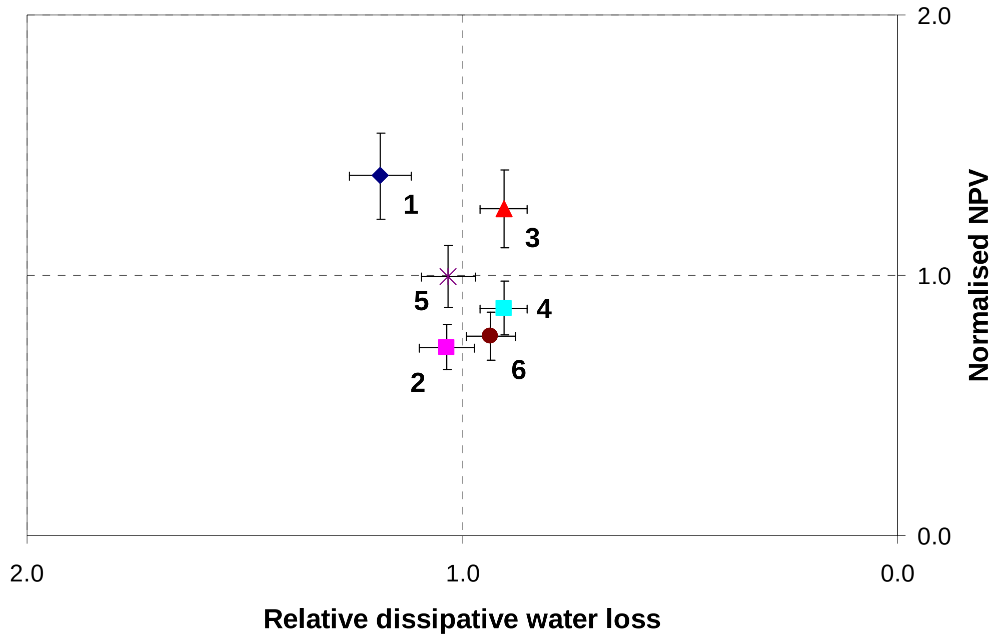

Figure 45: Relative graphical (normalised) eco-efficiency of gold tailings dewatering process alternatives

The above figure incorrectly suggests that option 3 is the most preferable option (since it is the only option in quadrant II), with option 2 being the least preferable (in quadrant IV). However, the above discussions have shown that the alternatives using high-rate thickeners (i.e. options 2, 4 and 6) are the most eco-efficient flowsheets. Furthermore, Figure 45 seems to suggest that set A (i.e. options 1, 3 and 5) have superior economic performance to that of set B. It is therefore apparent that the normalisation step performed above is insufficient in providing meaningful results, and therefore that the problematic nature of negative economic performance values still persists. An additional step in the analysis is therefore required. This incorrect interpretation can be rectified by rotating the economic performance values about the unity line of economic performance, which would therefore proportionally allocate the relative performance weights appropriately. This rotation is appropriate to the *relative* performance values now considered, and is different to the vertical shift about the x-axis that would be required if absolute values were considered, as postulated in Chapter 3. The results of this correction are shown in Figure 46.

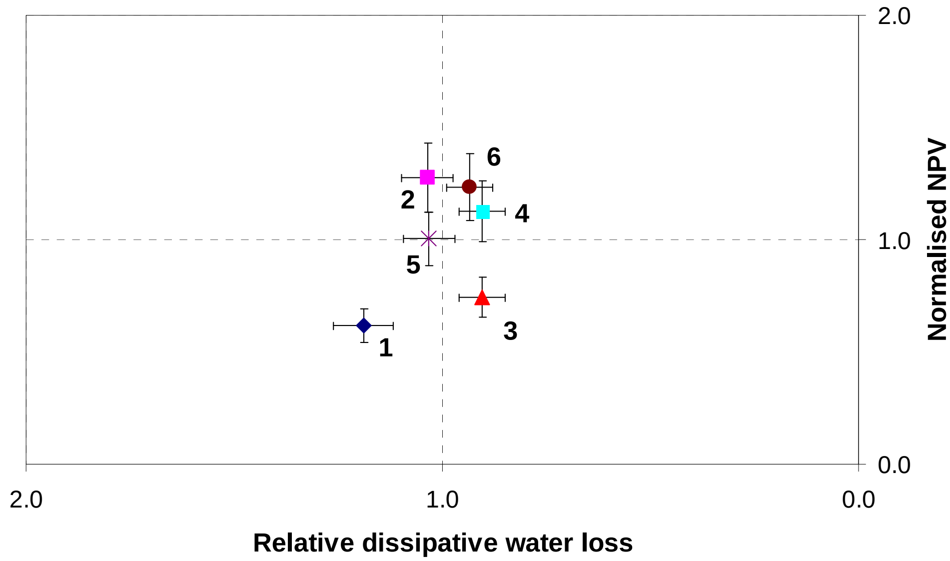

Figure 46: Relative graphical (normalised) eco-efficiency of gold tailings dewatering process alternatives (corrected)

The above performance depiction is now consistent with the absolute performance representations in Figure 41, Figure 42 and Figure 44, showing option 6 and option 4 to be the most eco-efficient dewatering circuit alternatives. This critique therefore demonstrates that the classical definition of eco-efficiency as used in this thesis and *as defined* by the WBCSD (2000) is not suitable for process design decision situations where a net economic performance loss is attained; additional analytical steps are required. Even with standard normalisation techniques (Saling *et. al.*, 2002), a careful differentiation between absolute and relative performance values (and how these in turn translate to the design decision space) is required to ensure meaningful results. The analysis therefore validates the need for the application of eco-efficiency (in numeric or graphical forms, and in absolute or relative terms) to be relevant to the decision context framing the design situation and the requisite analytical approach.

5.3.5 Distinguishability analysis

In validating claims made above, the results of the distinguishability analysis performed in this case study are shown in Figure 47 below, as distinguishability indices after Basson (2004). All indices and thresholds have been included in Appendix C.3.

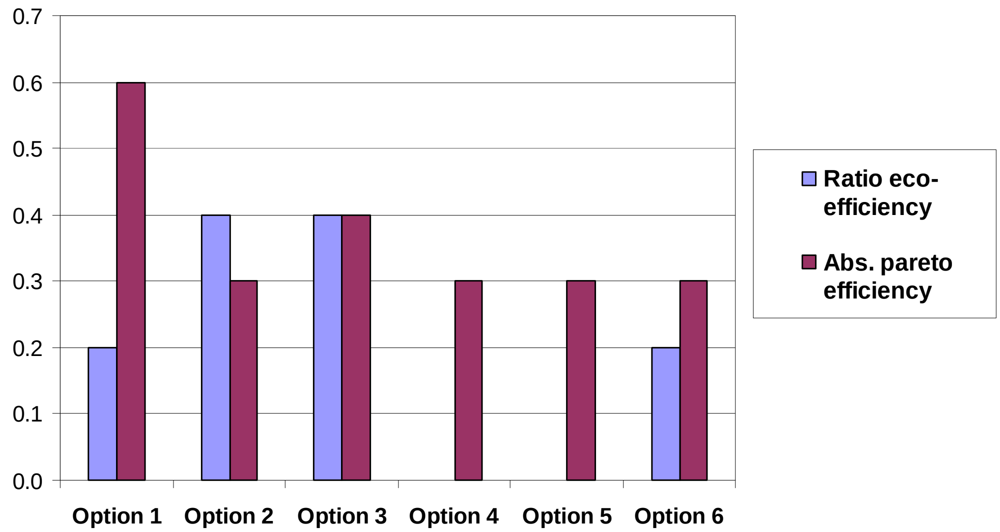

Figure 47: Ratio and absolute graphical eco-efficiency aggregated distinguishability indices in operational design

The above figure shows that four out of the six options have lower distinguishability scores for numeric eco-efficiency indicators than for graphical representations of economic and environmental performance. This suggests that eco-efficiency indicators still compound uncertainty even at relatively low levels of data uncertainty. This observation thus implies that even at operational levels of process design, where the quality (and quantity) of data available is much improved, the communication of performance information by numeric eco-efficiency indicators is less reliable than graphical depictions of the same data. It can also be observed from Figure 44 that while complete distinguishability is achieved between the economic and environmental performance of the two sets of dewatering circuits employing the different types of thickeners (i.e. set A and set B), options 2, 4 and 6 still display a considerable level of overlap. This therefore reinforces the conclusion that the claims regarding the discrepancies in eco-efficiency described above need to be made in light of distinguishability. In a real-time process design analysis, it would therefore be prudent to explore further opportunities for reducing uncertainty in the performance data to fully validate such conclusions.

5.3.6 Sensitivity analysis

In the tactical design case study in Chapter 4, the extent to which eco-efficiency indicators can be meaningfully linked to well-known technical process design parameters was briefly explored. This exercise has been repeated in the operational design decision context and is presented in this section. Supporting data is available in Appendix D.2.

The importance of the feed solids concentration in the performance of dewatering circuits is well accepted in the minerals process design literature (Mwale *et. al.*, 2005; Bedell *et. al.*, 2002). As such, the relationship between the economic and environmental performance values of the six dewatering circuit options and the circuit feed solids concentration has been investigated using a sensitivity analysis shown in Figure 48 below to highlight differences in performance between conventional and high-rate thickening. Option 5 and option 6 were used as examples of conventional and high-rate thickening, respectively.

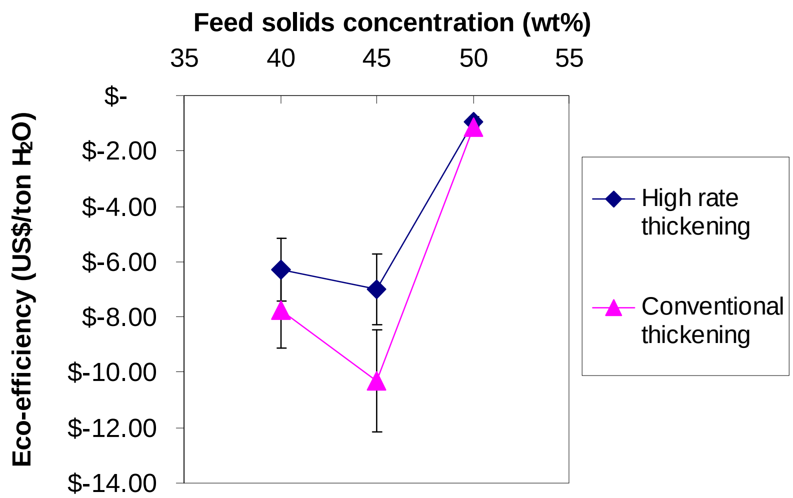

Figure 48: Sensitivity of the eco-efficiency indicator to the circuit feed solids concentration for option 5 and option 6

The above sensitivity analysis shows that the eco-efficiency of both circuits decreases considerably at feed solids concentrations lower than 50%. This is expected, given that an independent analytical study on the tailings properties of the GPP operation determined the ideal feed solids concentration for optimal settling conditions as approximately 49.7% (Resolute Tanzania Limited, 1999). The advantage of the high-rate thickening circuit over the conventional thickening circuit is also evident, with the high-rate option demonstrating a higher eco-efficiency for feed solids concentrations below 50%. This sensitivity analysis therefore further supports the claim made in Chapter 4 (for the same technology), maintaining that eco-efficiency indicators can be used to represent and explore meaningful trade-offs between technical and economic-environmental performance information. This ability to link these two sets of performance information may be particularly useful in operational design decision contexts such as in this case study, where the number of technical process variables governing performance criteria included in the design basis becomes considerably reduced relative to tactical design, therefore reducing the computational effort required by multicriteria decision analysis tools.

5.4 Summary

In this chapter, the application of eco-efficiency in an operational design decision context has been interrogated, using a gold tailings dewatering circuit design as a case study. In investigating the proposed techno-economic and environmental advantages of high rate thickening over conventional thickening, two possible sets of solutions in the design space were extracted, depending on whether a technological step-change or an incremental performance improvement was desired. The case study results have shown that while eco-efficiency indicators have communicated real performance improvement opportunities for some process design alternatives, even at operational levels of process design, where the quality (and quantity) of data available is much improved, the communication of performance information by eco-efficiency is less reliable than graphical depictions of the same data. Eco-efficiency indicators as defined by the WBCSD also proved to be unsuitable for application in design cases where economic losses are anticipated unless further analytical steps are performed. Inconsistencies between numerical and graphical representations of eco-efficiency performance also highlighted the danger in misguidedly using generic eco-efficiency indicators as all-encompassing decision-making tools during process design. Nonetheless, it has been demonstrated that eco-efficiency indicators represent a potentially powerful approach to linking economic-environmental performance information to technical process variables with which trade-offs can be explored also in operational design.

In the next chapter, the findings in this chapter are compared to observations made from the tactical design case study presented in Chapter 4. Thereafter, the salient findings, conclusions and final recommendations from the research are presented.

CHAPTER 6

Discussion and Conclusions

Having presented the economic and environmental performance assessment results for the two process design decision contexts investigated in Chapter 4 and Chapter 5 above, it is now of interest to compare these case studies with reference to the key research questions presented in Chapter 1 and developed in Chapters 2 and 3 of this thesis. This comparison forms the basis for discussing and interrogating the research hypotheses, formulating key conclusions and proposing further recommendations based on the research.

6.1 Comparison of the case studies

6.1.1 Discussion of the case studies

During the development of the research hypothesis in Chapter 3, a case was made for the application of eco-efficiency indicators in process design in tactical and operational decision contexts. It is useful to recall that the case studies used to explore this proposition were selected based on different *decision objectives*: in the tactical process design case study in Chapter 4, a preliminary process selection was desired as the objective of the design procedure to screen a suite of available copper processing technologies for the most desirable for further flowsheet development. By contrast, performance improvement given a set of constraints was sought in the operational process design case study presented in Chapter 5. Comparing the case studies based on these different decision objectives may therefore provide a useful starting point for framing some key characteristics of the corresponding decision contexts that are explored in this thesis. This comparison is thus provided below.

The manner in which the environmental impact categories were analysed for each of the two case studies has highlighted how eco-efficiency indicators offer flexibility in allowing for the consideration of only those environmental impacts that are relevant to the decision objective. Given the early phases of the design procedure in the copper beneficiation case study (and that a greenfield design was sought), the decision on process selection needed to be based on the consideration of *all* relevant environmental impacts. On the other hand, dissipative water loss was the dominant environmental impact that was relevant to the decision objective framing the gold dewatering circuit design case study. Eco-efficiency indicators can therefore be defined for single or multiple environmental impacts within an environmental performance analysis exercise. However, the copper beneficiation case study also showed a weakness in the use of eco-efficiency indicators in multiple-impact analyses due to the inherent inability of these indicators to prioritise which environmental impacts *are* relevant to the decision objective. As the case study has shown, these are chosen at the decision maker's discretion, with *a priori* knowledge or industry practices as underlying assumptions for selection. This lack of guidance also occurs at more detailed levels of analysis, as noted when eco-toxicity and resource depletion impacts were considered, where the substances contributing to the adverse impacts were somewhat arbitrarily chosen based on their concentrations or abundance in the original resource. Eco-efficiency analyses therefore need to be subordinate to more rigorous methodologies for the selection of decision objective-relevant environmental impacts at the onset of the analyses. Environmental risk-based approaches such as those motivated by Hansen (2004) and Broadhurst (2007a) may be promising in this regard.

Further strengths and limitations of eco-efficiency indicators relative to the decision objective were also evident from the economic performance assessments made in the case studies. Eco-efficiency indicators were noted to cope relatively well with decision situations where a net *positive* economic benefit was expected, such as in the copper beneficiation case study (when uncertainty limitations are not considered). However, the gold dewatering circuit design case study demonstrated that eco-efficiency indicators are unsuitable (at least in their classical ratio format) for decision situations where a net economic *loss* was expected. This comparison therefore suggests that the use of eco-efficiency indicators in performance-improvement or retrofit design procedures should be very carefully applied, if not best avoided. This assertion is re-visited shortly, when data quality considerations are also taken into account.

Despite the contrast between the above mentioned decision situations, it must be emphasised that in both case studies, the manner in which the economic benefit (i.e. numerator term of the indicators) is defined still fails to capture the hidden or externalised costs that exist due to both spatial and temporal effects that are typically not included in economic performance analyses, such as closure and rehabilitation costs, as well as costs associated with off-site environmental impacts (e.g. environmental damage from power generation). This shortcoming arises from a still persistent lack of data to describe these impacts and effects adequately and appropriately for meaningful decision making. Recent efforts in the development of environmental performance analysis tools that incorporate indirect impacts (such as life cycle assessment) have not yet been extended sufficiently to economic performance analyses. For example, other than NPV discounting (whose adequacy as an adequate methodological tool for a true economic performance reflection is being increasingly questioned within the environmental sciences – e.g. Boger and Hart (2008)), indirect temporal and spatial effects are hardly incorporated in economic analyses carried out during minerals process design. As a result, the system boundary inconsistencies that arise between economic and environmental data are exacerbated in performance indicators that combine these information sets into metrics, such as eco-efficiency indicators. This therefore limits their ability to respond adequately to process design decision objectives. Further development of analytical tools that increase the ease with which such indirect environmental impacts can also be economically quantified is therefore needed if eco-efficiency indicators are to provide internally consistent performance information for process design decision makers.

While the discussion thus far has highlighted some important system boundary considerations relative to the decision context, the case studies have not been explicitly compared in terms of uncertainty in process data as an element of the decision context. In doing so, it can be recalled that in the two preceding chapters, the eco-efficiency performance of process design alternatives was analysed based on the concept of distinguishability. Case study results indicated that eco-efficiency indicators compound uncertainty in economic and environmental performance information relative other more orthodox ‘graphical’ approaches. This was observed in both tactical and operational design decision contexts, underlining that this still occurs at even at relatively low levels of data uncertainty (such as in operational design decision contexts). In the tactical design decision context, the distinguishability analysis demonstrated that a ‘normalised’ graphical eco-efficiency interpretation should be preferred over numeric eco-efficiency, since this representation results in more distinguishable performance. This was also observed in the operational design decision context where, even with much lower levels of uncertainty, graphical Pareto-type eco-efficiency performance representations exhibited stronger distinguishability than numeric eco-efficiency indicators. Some important inconsistencies between numeric eco-efficiency indicators and graphical Pareto-based performance representations were also observed in this decision context, pointing to the potential for incorrect interpretation of numeric eco-efficiency indicators, particularly in the presence of uncertainty. The uncertainty analyses in these case studies have thus demonstrated that numeric eco-efficiency indicators compound uncertainty in performance information rather than reduce it, thus making them less desirable to graphical eco-efficiency performance representations when considered in this light.

Nonetheless, sensitivity analyses in both case studies have highlighted an important advantage in the application of numeric eco-efficiency indicators to minerals process design: expressing the economic and environmental performance information of process design alternatives as a function of process variables or parameters that are key *technical* performance criteria in the design basis. In the case of the tactical design decision context, the four eco-efficiency indicators analysed were varied according to the copper ore grade and metal production rate, while the dissipative water loss eco-efficiency indicator was varied as a function of the tailings feed solids concentration in the case of the operational decision context. This is of particular importance within the minerals and primary metals industries, where the concept of ‘performance’ in process design is still largely interpreted in techno-economic terms and the design basis is still dominated by technical process parameters (e.g. Scott, 2002), though these business models and views are changing (Petrie, 2007). However, given the relatively large size of design bases (key data such as ore grade, production rates, target resource and raw material recoveries, reactor and separator efficiencies etc. are typically included in a design basis), eco-efficiency indicators offer an opportunity for a richer communication of performance information: since many of these technical parameters are fundamental performance criteria, this approach explores their influence on the ‘macro’ or ‘systemic’ economic-environmental performance indicators, further strengthening the fundamental-systemic understanding of these design alternatives as motivated in this thesis through the systems approach. In an industry notorious for resistance to change and yet where practical solutions to environmental problems are held in high esteem (Broadhurst *et al.*, 2006), eco-efficiency indicators thus represent a practical and potentially powerful approach to linking economic-environmental performance information with technical process variables with which meaningful trade-offs can be explored during process design.

6.1.2 Towards a holistic design-for-environment decision support framework: An integrated approach for decision analysis tools

In reviewing the above discussion, the potential for applying numeric eco-efficiency analyses to different decision contexts and the ability of these analyses to meaningfully link economic-environmental performance criteria to key process variables in the design basis therefore emerge as two key strengths of eco-efficiency indicators. However, the inability of eco-efficiency indicators to generically guide the selection of appropriate environmental impacts for the analysis, their tendency to depend on (and encourage the use of) economic process data that excludes indirect impacts and their poor management of uncertainty are considerable shortcomings. These strengths and limitations make a strong case for the need to incorporate the use of eco-efficiency indicators into an environmental performance analysis framework that *includes* other process design decision analysis tools, rather than to use them as a stand-alone tool. Such a framework would provide useful ‘consistency checks’ for these indicators while retaining their flexibility to decision situations. It would also encourage the development of a more holistic approach to conducting environmental performance analyses in minerals process design by elucidating and strengthening the links between environmental decision analysis tools that can by symbiotically applied at various stages of any process design procedure. Some of the limitations of these indicators may therefore be mitigated in this manner. For example, environmental risk-based approaches may be used to guide a ‘first principles’ selection of environmental impacts to prioritise in the eco-efficiency analysis. Distinguishability analyses may also be used as a basis for incorporating uncertainty reduction techniques more rigorously into process design, such as those described by Basson (2004). Sensitivity analyses have also demonstrated how an awareness of system boundaries can contribute to better accounting of environmental impacts while remaining relevant to objectives of the design basis. An example of how such a framework might be mapped based on the arguments made in this thesis is shown in Figure 49 below.

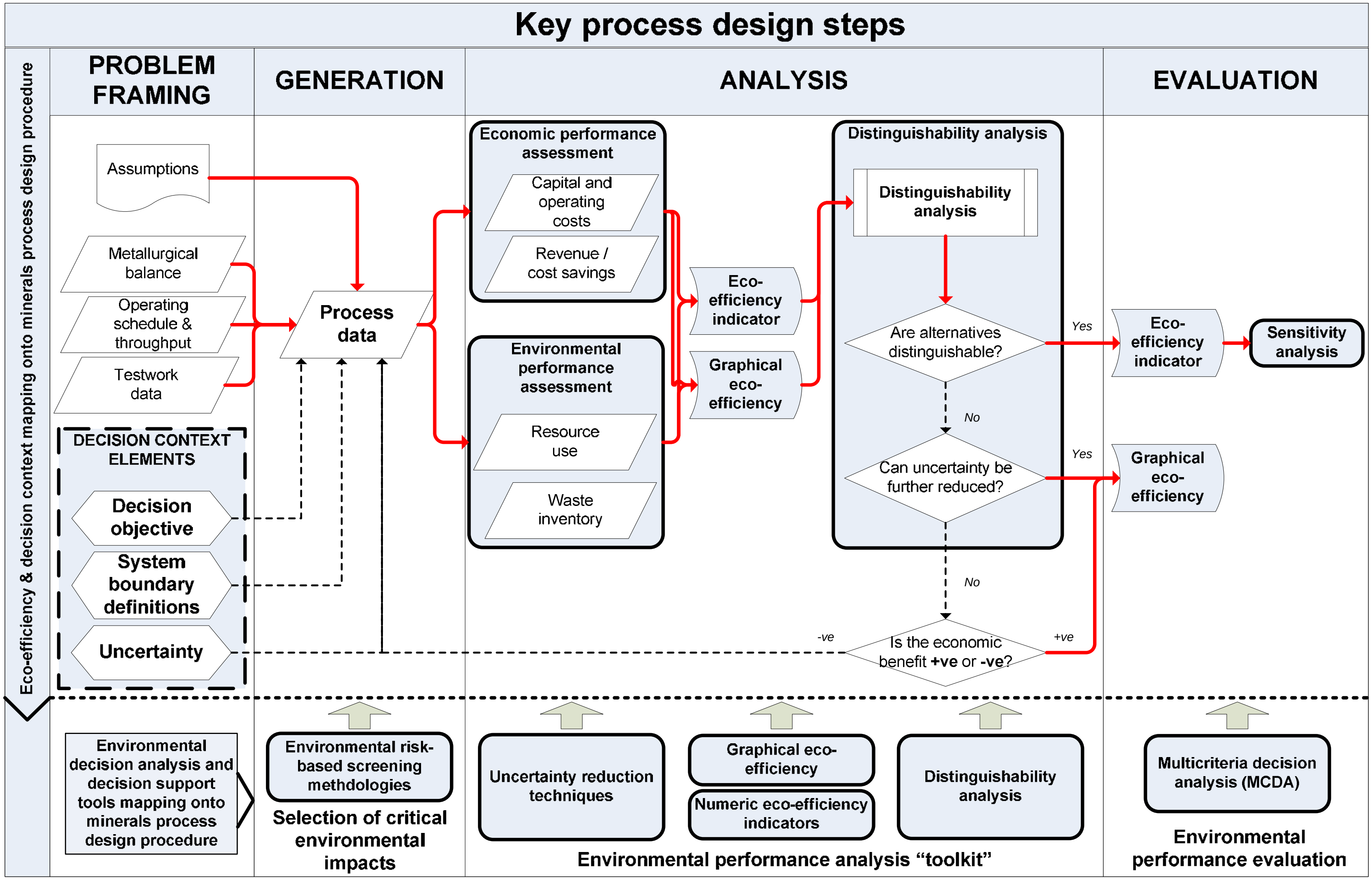

Figure 49: A proposed basis for an environmental decision analysis framework to eco-efficiency indicator development

6.2 Thesis appraisal and conclusions

The preceding discussions in this chapter have compared the findings from the application of eco-efficiency indicators in one tactical and one operational design decision context case study. To conclude this thesis, these findings are further distilled in this section in light of the initial motivation for the research, the objectives of this thesis presented in Chapter 1 and the hypotheses developed in Chapter 3. Recommendations to minerals process design industry practitioners and researchers are then offered based on these conclusions.

6.2.1 Motivation for conducting the research

Eco-efficiency has been proposed by the World Business Council for Sustainable Development as an environmental performance indicator framework that contributes to sustainability by assisting corporate decision makers improve the environmental performance of their operations and processes, while also extracting additional economic value. Furthermore, the recent engagement of the minerals and primary metals industries with sustainability has highlighted the need for a systems-based approach to developing and applying the requisite tools and methodologies. However, the use of eco-efficiency indicators as environmental performance metrics to meaningfully guide decision making during process design in the minerals beneficiation sector had yet to be explored. A knowledge gap therefore existed between the use of eco-efficiency and decision making during minerals process design, where significant opportunities for environmental performance improvement exist. This knowledge gap presented a need to explore how systems-based approaches could be applied in developing eco-efficiency indicators as a set of sustainability performance analysis metrics to generate the requisite data for guiding more environmentally sustainable decision making during process design.

6.2.2 Objectives of the research

Based on the above motivation, this thesis therefore sought to assess the strengths and limitations of eco-efficiency indicators as performance metrics in guiding environmentally sustainable decision making during minerals process design. In particular, the following key questions guiding the research were defined:

a) How can eco-efficiency indicators be used to describe the environmental and economic performance of minerals process design alternatives?
b) How can eco-efficiency indicators assist in the framing of decision objectives for minerals process design?

c) What relationships exist between the eco-efficiency indicators as performance metrics, the underlying requisite data and the decision objectives desired during minerals process design?

This objective was achieved firstly by offering a more rigorous definition of eco-efficiency indicators for the minerals and primary metals industries based on indicators currently offered in the literature. Based on a case study research design, two case studies were then selected to reflect typical but different decision situations encountered within minerals process design, informed by the concept of design decision contexts. These case studies represented the *tactical* and *operational* decision contexts in minerals process design. For each case study, eco-efficiency indicators were defined, computed and were compared to more traditional graphical representations of the economic and environmental performance of process design alternatives considered, defined in absolute (Pareto-type) and relative (normalised) terms. Results were validated with distinguishability analyses, and the sensitivity of these indicators to some key technical process design parameters was also investigated.

6.2.3 Key conclusions from the research

The major conclusions of this thesis can be summarised as follows.

1) Eco-efficiency indicators can be successfully applied in some but not all minerals process design situations, based on scientific rigour and precise methodologies.

The selected case studies were successful in demonstrating how eco-efficiency indicators can be used to describe the environmental and economic performance of minerals process design alternatives. In the tactical design context, *greenhouse gas emissions (global warming potential)*, *water consumption*, *aquatic eco-toxicity* and *resource depletion* eco-efficiency indicators were defined and applied to characterise the economic and environmental performance of three different copper beneficiation and processing routes. In the operational design context, a *dissipative water loss* eco-efficiency indicator was defined and used to assess the performance of six tailings dewatering circuit design alternatives in an existing gold ore processing facility. These definitions and applications have therefore contributed towards a scientifically rigorous and methodologically precise development of eco-efficiency indicators specifically for the minerals and primary metals industries. It must be noted though that the numeric ratio-based eco-efficiency indicator provided misleading results in the case of a net economic loss, whilst graphical methods remained useful. Also, the use of such indicators must be subordinate to the application of requisite methods in the *selection* and *prioritisation* of environmental impacts that are relevant to each design situation.

2) The decision objectives, quality of process performance information and system boundary definitions are key elements of the decision context which must be explicitly addressed in problem framing during minerals process design.

Given the above discussions on the scope of applicability of eco-efficiency indicators, the research design and the case study results have demonstrated that the concept of the *decision context* provides an insightful manner for describing different decision situations that can be encountered in minerals process design. Key differences between the tactical and operational design decision contexts were deduced, therefore offering insights into how process design problems need to be framed. These included:

- Differences in *decision objectives* (preliminary process selection for greenfield design vs. performance improvement in an existing operation),
- Differences in the *quality of process performance information* required for decision making (higher vs. lower degrees of uncertainty), and
- Differences in the requisite *resolution and scope of system boundaries* defined for the environmental performance analyses (wide vs. narrow system boundaries, including the inclusion or exclusion of indirect environmental impacts).

Therefore, in addition to decision objectives, the case study results have confirmed that the quality of process performance information and definition of system boundaries are key elements of the decision context that need to be brought to the forefront during the problem framing step of the design procedure when environmental performance analyses are to be undertaken.

3) Despite data uncertainty and system boundary limitations, eco-efficiency indicators show value in relating systems-level environmental and economic performance information to fundamental technical performance criteria

The above differences were further improved when the (numeric) eco-efficiency indicators were compared to graphical approaches to eco-efficiency performance depiction. Both Pareto-type and normalised graphical eco-efficiency exhibited higher distinguishability in the economic and environmental performance of design alternatives when compared to numeric eco-efficiency indicators, therefore highlighting the relatively poor management of uncertainty associated with eco-efficiency indicators. These results were also confirmed with distinguishability analyses. Furthermore, inconsistencies noted between eco-efficiency indicators and graphical performance representations highlighted that these indicators are not suitable for application in design decision situations where a net economic loss is achieved. Eco-efficiency indicators were also shown to depend on process data that excludes the consideration for indirect impacts (both environmental and financial). Nonetheless, the sensitivity analyses conducted for each case study have demonstrated that eco-efficiency indicators can meaningfully link economic-environmental performance criteria to key technical process variables and parameters in the minerals process design basis – an important strength of these indicators and a key contribution of this thesis.

This thesis has therefore provided some useful insights into the use of systems-based approaches to mapping important relationships between environmental performance analysis tools and their metrics, the amount and quality of the underlying requisite data and the decision context in which these tools and metrics are applied during minerals process design. In this manner, the three key questions posed in Chapter 1 of this thesis have been successfully answered. The thesis has also directly contributed to the research efforts of the University of Cape Town's Minerals-to-Metals research initiative, forming a solid foundation for a deeper exploration of how integrated approaches incorporating other environmental decision analysis tools can be developed for the minerals and primary metals industries.

6.2.4 Validation of the hypothesis

The hypotheses that were proposed for this thesis as developed in Chapter 3 can now be revisited.

1) **Eco-efficiency indicators can meaningfully communicate the environmental and economic performance of design alternatives in minerals process design.** *(Hypothesis 1)*

Some important limitations of eco-efficiency indicators were uncovered during the case study investigation. Whilst the use of eco-efficiency indicators in sensitivity analyses could add significant value to the design of minerals beneficiation processes, their use should be restricted to cases of positive economic value and positive environmental damage, and uncertainty propagation must be considered. Graphical depictions of environmental and economic performance have been shown in general to be more trustworthy. This hypothesis has therefore not been universally validated.

2) **The use of eco-efficiency indicators to inform decision making in minerals process design needs to be governed by the decision context.** *(Hypothesis 2)*

The results from the case studies in Chapter 4 and Chapter 5 have highlighted the key systemic differences that govern how eco-efficiency indicators should be applied in tactical and operational design decision contexts, as summarised in the preceding section. The use of the decision context concept as a platform for integrating various decision analysis and support tools for more environmentally sustainable decision making has also been demonstrated. The hypothesis is therefore considered valid and is not rejected.

6.3 Recommendations

Based on the above conclusions, a summary of recommendations to minerals industry practitioners and researchers has been provided below.

6.3.1 Recommendations for industry process design engineers

The following recommendations can be made for process designers in the minerals industry:

1) Eco-efficiency indicators need to be used in a manner that is subordinate to the application of requisite methods in the *selection* and *prioritisation* of environmental impacts relevant to each design situation.
2) Eco-efficiency indicators should be used in cases of positive economic value and positive environmental damage, and uncertainty propagation must be explicitly considered during the analysis. Otherwise, graphical representations of eco-efficiency performance should be employed.
3) Sensitivity analyses linking technical design parameters to eco-efficiency indicators should be used where multicriteria decision analysis tools are preferred, should the situations reflected by point 1 and point 2 above be applicable.

6.3.2 Recommendations for further research

In addition to the recommendations on the application of eco-efficiency indicators for minerals process design made above, the following recommendations are offered regarding a need for further research.

1) The relationship between eco-efficiency indicators and technical process parameters is regarded as holding significant value towards environmental multicriteria decision making in minerals process design, but has been only superficially explored in this thesis. A deeper exploration is therefore needed.
2) Linkages between environmental risk-based approaches to generalised screening of environmental impacts (for ensuring optimum relevance to each process design context) and eco-efficiency environmental impact categories should be further explored, particularly when eco-toxicity and resource depletion impacts are considered.
3) Distinguishability analyses performed in this thesis should be extended to identify areas of uncertainty reduction for a meaningful comparative evaluation of process design alternatives (should distinguishability be a limiting factor to decision making, as has been the case in this thesis).
4) The manner in which eco-efficiency performance analyses can be integrated into modern technical process design software and virtual platforms is an important research question

that deserves further research. Data quality and system boundary definitions highlighted in this thesis could be used as a basis for such an interrogation.

5) Pending practical limitations, the conclusions made in this thesis can be further strengthened by including more case studies in future research for a richer analysis.

References

Alexander, B., Barton, G., Petrie, J. and Romagoli, J.; **2000**. Process synthesis and optimisation tools for environmental design: methodology and structure, *Computers and Chemical Engineering*, 24: 1195-1200.

Ally, M., Berry, J., Dole, L., Ferrada, J. and Van Dyke, V.W.; **2001**. *Economic Recovery of Byproducts in the Minerals Industry: Report No. ORNL/TM-2001/225*, US Department of Energy, Washington DC, the United States, pp 1-7.

Anglo American; **2007**. *Anglo American Plc Annual Report 2007: Report*, London, UK.

Ashton, P.J., Love, D., Mahachi, H. and Dirks, P.H.G.M.; **2002**. *An overview of the impact of mining and minerals processing operations on water resources and water quality in the Zambezi, Limpopo and Olifants Catchments in Southern Africa: Report*, MMSD Southern Africa, Pretoria, South Africa, pp 3-61

Ayres, R, Ayres, L and Råde, I.; **2002**. *The Life Cycle of Copper, it's co-products and by-products: Report*, London, UK

Azam, S.; **2004**. Thickening of mine waste slurries, *Geotechnical News*, 12: 41-43.

Azapagic, A. and Perdan, S.; **2000**. Indicators for sustainable development for industry: A general management framework, *Transactions of the Institution of Chemical Engineers, Part B*, 78: 243-261.

Azapagic, A.; **2003**. Systems approaches to corporate sustainability: A general management framework, *Transactions of the Institution of Chemical Engineers, Part B*, 81: 303-316.

Azapagic, A.; **2004**. Developing a framework for sustainable development indicator for the mining and minerals industry, *Journal of Cleaner Production*, 12: 639-662.

Basson, L. and Petrie, J.; **2001**. *A Roadmap for Decision-making in Different Decision Contexts*, **6th World Congress of Chemical Engineering 2001**, Melbourne, Australia; Melbourne, VA: AusIMM, pp 1-4, 8-10

Basson, L. and Petrie, J.; **2007**. A critical systems approach to decision support for process engineering, *Computers and Chemical Engineering*, 31: 876-888.

Basson, L.; **2004**. *Context, Compensation and Uncertainty in Environmental Decision Making*, PhD Thesis, **University of Sydney**, Sydney, Australia, pp 1-4 – 1-13, 2-1 – 2-18, 3-17 – 3-19, 5-1 – 5-30.

Basu, A. and van Zyl, D.; **2006**. Industrial ecology framework for achieving cleaner production in the mining and minerals industry. *Journal of Cleaner Production*, 14: 299-304.

Batterham, R.; **2006**. Sustainability - The next chapter, *Chemical Engineering Science*, 61: 4188-4193.

Baumann, H. and Cowell, S.J.; **1999**. An Evaluative Framework for Conceptual and Analytical Approaches Used in Environmental Management, *Greener Management International*, 26: 109-122.

Bedell, D., Slottee, S., Parker, K. and Henderson, L.; **2002**. The thickening process. In: Jewell, R. J., Fourie, A. B. and Lord, E. R. (eds): *Paste and thickened tailings – A guide*. Nedlands, WA: The Australian Centre of Geotechnics, pp 49-79.

Bekker, S.; **2008**. *Personal communication*.

Belton, V. and Stewart, T.; **2002**. *Multiple Criteria Decision Analysis: An Integrated Approach*, Dordrecht: Kluwer Academic Publishers, pp 1-11

Biegler, L., Grossmann, I.E. and Westerberg, A.; **1997**. *Systematic methods for chemical process design*, Upper Saddle River, NJ: Prentice Hall, pp 1-2 – 1-5, 2-8 – 2-9, 4-15.

Biswas, A. and Davenport, W., 1994. Extractive metallurgy of copper, 3rd ed., Elsevier Science: Oxford.

Boger, D.V. and Hart, B.C.; **2008**. *Making an Unsustainable Industry More Sustainable*, **2008 International Seminar on Paste and Thickened Tailings**, Gaborone, Botswana; Nedlands, WA: The Australian Centre for Geomechanics, pp 3-5.

Bonano, E.; **1995**. Decision Making under Uncertainty and Risk Assessment. In R. Deshotels and R. Zimmerman (Ed.), *Cost-Effective Risk Assessment for Process Design,* New York, NY: McGraw-Hill, pp 11-16

Brackebusch, F. W.; **1994**. The basics of paste backfill systems, *Mining Engineering*, 11: 1175-1178.

Brattebø, H., Røine, K., Eik, A., Bohne, R. and Bergsdal, H.; **2006**. *Sound Eco-Efficiency Approaches in Recycling Systems – Implications of Complexity, Scale and Dynamics*, **2nd International *Conference* on Quantified Eco-Efficiency Analysis for Sustainability 2006**, Egmond aan Zee, Leiden: Leiden University, pp 5.

Brierley, C.L. and Brierley, J.A.; **1999**. Bioheap Processes – Operational Requirements and Techniques. In Jergensen, G.V. (ed.), *Copper Leaching, Solvent Extraction and Electrowinning Technology*, Littleton, CO: SME, pp 17-29

Broadhurst, J., Petrie, J. and von Blottnitz, H.; **2006**. *Understanding element distribution during primary metal production*, **Green Processing 2006**, Melbourne, Australia; Melbourne, VA: AusIMM.

Broadhurst, J.; **2007a**. *Predicting the characteristics of solid wastes from the resource-based industries: A focus on primary copper production*, PhD Thesis, **University of Cape Town**, Cape Town, South Africa, pp 1-1 – 1-13, 2-1, 4-1.

Broadhurst, J.; **2007b**. *Personal communication*.

Bulatovic, S. and Wyslouzil, **1999**. Development and application of new methodology for the treatment of massive sulphide ores case study – Faro lead/zinc concentrator, Yukon, *Minerals Engineering*, 12 (2): 129-145.

Cabezas, H., Bare, J. and Mallick, S.; **1999**. Pollution prevention with chemical process simulators: the generalized waste reduction (WAR) algorithm, *Computers and Chemical Engineering*, 23: 623-634.

Cano-Ruiz, J. and McRae, G.; **1998**. Environmentally conscious chemical process design, *Annual Review of Energy and the Environment*, 23: 499-536.

Cha, K., Lim., S. and Hur, T.; **2007**. Eco-efficiency approach for global warming in the context of Kyoto Mechanism, *Ecological Economics* (Article in press).

Chakraborty, A., Purkarthofer, K. and Linninger, A.; **2004**. Conceptual process design based on thermodynamic and economic insights, *Chemical Engineering and Processing*, 43: 625-640.

Checkland, P.; **1985**. Science and the systems movement. In J. Beishon and G. Peters (Ed.), *Systems Behaviour* pp. 26-43, London: Harper & Row.

Chen, H. and Shonnard, D.; **2004**. Systematic framework for environmentally conscious chemical process design: Early and detailed stages, *Industrial and Engineering Chemistry Research*, 43: 535-552.

Cisternas, L., Gálvez, E., Zavala, M. and Magna, J.; **2004**. A MILP model for the design of mineral flotation circuits, *International Journal of Mineral Processing*, 74: 121-131.

Clift, R.; **2006**. Sustainable development and its implications for chemical engineering, *Chemical Engineering Science*, 61: 4179-4187.

Copeland, A.; **2008**. *Personal communication*.

Dahlstrom, K. and Ekins, P.; **2005**. Eco-efficiency trends in the UK steel and aluminium industries, *Journal of Industrial Ecology*, 9 (4): 171-188.

Datastream; **2008**. Thomson Financials Statistical Database [online], Available from: http://www.datastream.com, Accessed 10/03/2008.

Dimian, A., Groenendijk, A., Kersten, S. and Iedema, P.D.; **1997**. Effect of recycle interactions on dynamics and control of complex plants, *Computers and Chemical Engineering*, 21 (suppl.): S291-S296.

Douglas, J.; **1988**. *Conceptual Design of Chemical Processes*, New York, NY: McGraw-Hill, pp 1-5 – 1-25, 5-8.

DWAF; **2007**. Development of Best Practice Guidelines for Water Resource Protection in the South African Mining Industry. *Report A3: Water Management in Hydrometallurgical Plants*, National Department of Water Affairs and Forestry, Pretoria, South Africa, pp 2-21.

Ehrenfeld, J.; **2005**. Eco-efficiency: Philosophy, Theory, and Tools, *Journal of Industrial Ecology*, 9 (4): 6-8.

Ekins, P.; **2005**.Eco-efficiency: Motives, Drivers and Economic Implications, *Journal of Industrial Ecology*, 9 (4): 12-14.

Elliott, A.D., Sowerby, B. and Crittenden, B.D.; **1996**. Quantitative environmental impact analysis for clean design, *Computers and Chemical Engineering*, 20 (S2): S1377-S1382.

Erkko, S., Melanen, M. and Mickwitz, P.; **2005**. Eco-efficiency in the Finnish EMAS reports: a buzz word? *Journal of Cleaner Production*, 13: 799-813.

ESKOM; **2006**. *ESKOM Annual Report 2006: Report*, Johannesburg, South Africa.

Frigge, F. and Hahn, T.; **2004**. Sustainable Value Added—measuring corporate contributions to sustainability beyond eco-efficiency, *Ecological Economics*, 48: 173-187.

Frischknecht, R., Jungbluth, N., Althaus, H-A., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M. and Wernet, G.; **2007**. *ecoinvent Data 2.0 (2007) – Overview and Methodology: Report No. 1*, Dübendorf, Switzerland.

Functowitz, S. and Ravetz, J.; **1990**. *Uncertainty and Quality in Science for Policy*, Dordrecht: Kluwer Academic Publishers.

Georgalli, G.A., Eksteen, J.J. and Reuter, M.A.; **2002**. An integrated thermochemical-systems approach to the prediction of matte composition dynamics in an Ausmelt nickel–copper matte converter, *Minerals Engineering*, 15: 909-917.

Giurco, D.P.; **2005**. *Towards sustainable metal cycles: The case of copper*, PhD Thesis, **University of Sydney**, Sydney, Australia, pp 7-44, 179-183, 188-224.

Giurco, D.P. and Petrie, J.G.; **2007**.; Strategies for reducing the carbon footprint of copper: New technologies, more recycling or demand management? *Minerals Engineering,* 20: 842-853.

Giurco, D.P., Petrie, J.G. and Stewart, M.; **2000**.; The role of LCA in performance assessment in minerals processing – a copper case study, **SWEMP 2000**, Calgary, Canada, pp. 1-6.

Goedkoop, M., Oele, M., de Schryver, A. and Vieira, M.; **2008**. *SimaPro Database Manual Methods Library: PRé Report*, Amersfoort, the Netherlands, pp 1-52.

Goodland, R. and Daly, H.; **1996**. Environmental sustainability: Universal and Non-negotiable, *Ecological Applications*, 6: (4): 1002-1017.

Grossmann, I. E., and Westerberg, A. W.; **2000**. Research challenges in process systems engineering. *AIChE Journal*, 46 (9): 1700-1703.

Grossmann, I. E.; **2004**. Challenges in the new millennium: Product discovery and design, enterprise and supply chain optimisation, global life cycle assessment. *Computers and Chemical Engineering*, 29: 29-39.

Hansen, Y.; **2004**. *Environmental impact assessment of solid waste management in the primary industries – A new approach*. PhD Thesis, **University of Sydney**, Sydney, Australia, pp. 2-1 – 2-42.

Herbst, J., Rajamani, R., Mular, A. and Flintoff, B.; **2002**. *Mineral Processing Plant/Circuit Simulators: An Overview*, **Mineral Processing Plant Design, Practice and Control Symposium 2002**, Vancouver, Canada; Littleton, CO: SME, pp 383-401.

Hilson, G. and Murck, B.; **2000**. Sustainable development in the mining industry: clarifying the corporate perspective. *Resources Policy*, 26: 227-238.

Hilson, G. and Nayee, V.; **2002**. Environmental management system implementation in the mining industry: a key to achieving cleaner production. *International Journal of Minerals Processing*, 64: 19-41.

Hilson, G.; **2000**. Barriers to the implementing cleaner technologies and cleaner production (CP) practices in the mining industry: A case study of the Americas. *Minerals Engineering*, 13 (7): 699-717.

Hilson, G.; **2001**. Putting theory into practice: How has the gold mining industry interpreted the concept of sustainable development, *Mineral Resources Engineering*, 10: (4): 397-413.

Hilson, G.; **2003**. Defining “cleaner production” and “pollution prevention in the mining context”. *Minerals Engineering*, 16: 305-321.

Huband, S., Tuppurainen, D., While, L., Barone,.L., Hingston, P. and Bearman, R.; **2006**. Maximising overall value in plant design, *Minerals Engineering*, 19: 1470-1478.

Hukkinen, J.; **2001**. Eco-efficiency as abandonment of nature, *Ecological Economics*, 38: 311-315.

Huppes, G. and Ishikawa, M.; **2005**. A framework for quantified eco-efficiency analysis, *Journal of Industrial Ecology*, 9 (4): 25-41.

ICME; **2001**. *Eco-efficiency and Materials: Foundation Paper*, Ottawa, Canada, pp 2-6.

ICMM; **2006**. *Participatory Planning near Golden Pride mine in Nzega District, Tanzania* [online], Available from: http://www.icmm.com/casestudy.php?rcd=42, Accessed 12/10/2007.

IIED ; **2001**. *Mining, Minerals and Sustainable Development Project: Breaking New Ground: Report*, London, UK, pp 2-13.

International Copper Study Group; **2008**. *World Copper Factbook 2007: Report*, ICSG, Lisbon, Portugal, pp 1-66.

ISO; **1999**. ISO 14031 - Environmental Management: Environmental Performance Evaluation Guidelines (E), Geneva: ISO, pp 1-40.

Jaspers, F.; **2007**. Case study research: Some other applications besides theory building Case study research: Some other applications besides theory building, *Journal of Purchasing and Supply Management*, 13: 210-212.

Jenkins, J.; Davenport, W.G., Kennedy, B. and Robinson, T.; **1999**. Electrolytic copper - leach, solvent extraction and electrowinning world operating data, **Copper 1999**, Phoenix AZ, the United States, pp 41-88.

Jergensen, G.V. (Ed.); **1999**. *Copper Leaching, Solvent Extraction and Electrowinning Technology*, Littleton, CO: SME, pp 1-109.

Jewell, R.J., Fourie, A.B. and Lord, E.R. (eds); **2006**. *Paste and Thickened Tailings – A Guide*, 2nd edition. Nedlands, WA: The Australian Centre for Geomechanics, pp 1-173.

Jollands, N.; **2006**. Concepts of efficiency in ecological economics: Sisyphus and the decision maker, *Ecological Economics*, 56: 359-372.

Kharel, G.P. and Charmondusit, K.; **2007**. Eco-efficiency evaluation of iron rod industry in Nepal. *Journal of Cleaner Production* (Article in press).

Korhonen, J.; **2007**. Special issue of the Journal of Cleaner Production, ‘From Material Flow Analysis to Material Flow Management’: strategic sustainability management on a principle level (Editorial), *Journal of Cleaner Production*, 15: 1585-1595.

Kuosmanen, T. and Kortelainen, M.; **2005**. Measuring Eco-efficiency of production with data envelopment analysis, *Journal of Industrial Ecology*, 9 (4): 59-72.

Kurtz, J., Jackson, L. and Fisher, W.; **2001**. Strategies for Evaluating Indicators Based on Guidelines from the Environmental Protection Agency's Office of Research and Development Ecological Indicators, *Ecological Indicators*, 1 (1): 49-60.

Liu, Y. and Spencer, S.; **2004**. Dynamic simulation of grinding circuits, *Minerals Engineering*, 17: 1189-1198.

Mäkinen, J.; **2006**. *Eco-efficient Solutions in the Finnish Metallurgical Industry*, **2nd International *Conference* on Quantified Eco-Efficiency Analysis for Sustainability 2006**, Egmond aan Zee, The Netherlands; Leiden: Leiden University, pp 1-42.

Marr, S.; **2003**. *Environmental considerations in the design of base metals refineries: Metals in riverine systems*, MSc. Thesis, **University of Cape Town**, Cape Town, South Africa, pp 23-152.

Matche Consultants; **2007**. *Matches' Process Equipment Cost Estimates* [online], Available from: http://www.matche.com/EquipCost/index.htm, Accessed 21/10/2007.

Mattson, C. and Messac, A.; **2003**. Pareto frontier based concept selection under uncertainty with visualization, *Optimization and Engineering*, 6: 85-115.

Mayer, E.; **2000**. Cake filtration theory and practice, *Chemical Engineering Journal*, 80: 233-236.

Mebratu, D.; **1998**. Sustainability and Sustainable Development: Historical and Conceptual Review, *Environmental Impact Assessment Review*, 18: 493-520.

Meggyes, T.; **2004**. Paste and Thickened Tailings Technology (Unpublished manuscript), Perth, WA: Federal Institute for Materials Research and Testing, pp. 2-5.

Mesarovic, M. and Takahara, Y.; **1975**. General Systems Theory: Mathematical Foundations. In R. Bellman (Ed.), *Mathematics in Science and Engineering* New York, NY: Academic Press, pp 12.

Metso Minerals; **2006**. *Metso in Filtration Brochure No. 1623-03-06-MPR/Sala*, Metso Minerals (AB), Helsinki, Finland, pp 3.

Michelsen, O., Fet, A.M. and Dahlsrud, A.; **2006**. Eco-efficiency in extended supply chains: A case study of furniture production, *Journal of Environmental Management*, 79: 290-297.

Mickwitz, P., Hyvättinen , H. and Kivimaa, P.; **2008**. The role of policy instruments in the innovation and diffusion of environmentally friendlier technologies: popular claims versus case study experiences, *Journal of Cleaner Production*, 16S1: S162-S170.

Milhelcic, J., Crittenden, J., Shonnard, D., Hokanson, D., Zhang, Q., Chen, H., Sorby, S., James, V., Sutherland, J. and Schnoor, J.; **2003**. Sustainability Science and Engineering: The Emergence of a New Metadiscipline, *Environmental Science & Technology*, 37: 5314-5324.

Morgan, M. and Henrion, M.; **1990**. *A Guide to Dealing with Uncertainty in Quantitative Risk and Policy Analysis*, New York, NY: Cambridge University Press, pp 109.

Morrison, R. and Richardson, J.; **2002**. *JKSimMet: A simulator for analysis, optimisation and design of comminution circuits*, **Mineral Processing Plant Design, Practice and Control Symposium 2002**, Vancouver, Canada; Littleton, CO: SME, pp 442-460.

Mouton, J.; **2001**. *How to succeed in your masters and doctoral studies*, 1st ed., Pretoria: Van Schaik, pp 5-23.

Mudacumura, G., Mebratu, D. and Haque, S.; **2006**. *Sustainable Development Policy and Administration*, London: CRC Press, pp 200-272.

Mudd, G. ;**2007**. Global trends in gold mining: Towards quantifying environmental and resource sustainability, *Resources Policy*, 32 (1-2): 42-56.

Müller, K. and Sturm, A.; **2001**. *Standardized Eco-efficiency Indicators: Report No. 1: Concept Paper*, Basel, Switzerland, pp 1-11.

Mwakyusa, G.; **2007**. *Tailings dewatering circuit design for optimal water recovery*, MSc. Thesis, **University of Cape Town**, Cape Town, South Africa, pp 11-96.

Mwale, A.H., Musonge, P. and Fraser, D.; **2005**. The influence of particle size on energy consumption and water recovery in comminution and dewatering systems, *Minerals Engineering*, 18: 915-926.

Neimeijer, D. and de Groot, R.; **2008**. Framing environmental indicators: moving from causal chains to causal networks, *Environment, Development and Sustainability*, 10: 89-106.

Niemi, G. and McDonald, M.; **2004**. Application of ecological indicators, *Annual Review of Ecology and Evolution Systems*, 35: 89-111.

Nieuwlaar, E., Warringa, G., Brink, C. and Vermeulen, W.; **2005**. Supply curves for Eco-efficient environmental improvements using different weighting methods, *Journal of Industrial Ecology*, 9: (4): 85-96.

Norgate, T. E. & Rankin, W. J.; **2002**. *The Role of Metals in Sustainable Development*, **Green Processing 2002: International Conference on the Sustainable Processing of Minerals**, Melbourne, Australia, Melbourne, VA: AusIMM, pp 49-55.

Norgate, T. E. and Rankin, W. J.; **2000**. *Life cycle assessment of copper and nickel production*, **AusIMM MINPREX 2000**, Melbourne, Australia, Melbourne, VA: AusIMM, pp. 133-138.

Norgate, T. E., Jahanshahi, S. and Rankin, W. J.; **2007**. Assessing the environmental impact of metal production processes, *Journal of Cleaner Production*, 15: 838-848.

Notten, P.; **2002**. *Life-Cycle Inventory Uncertainty in Resource-Based Industries - A Focus on Coal-Based Power Generation*, PhD Thesis, **University of Cape Town**, Cape Town, South Africa, pp 2-1 – 2-16, 3-1, 4-1.

OECD ; **2001**. *OECD Environmental Indicators: Towards Sustainable Development. Organisation for Economic Co-operation and Development: Report*, Paris, France, pp. 2-148.

Patterson, A.J.C.; **2008**. *Personal communication*.

Peacey, J., Guo, X.J. and Robles, E.; **2003**. *Copper Hydrometallurgy – Current Status, Preliminary Economics, Future Direction and Positioning versus Smelting: Report*, Ontario, Canada, pp 205-222.

Petersen, J. and Dixon, D.G.; **2006**. Competitive bioleaching of pyrite and chalcopyrite, *Hydrometallurgy*, 83: 40-49.

Petrie, J., Cohen, B. and Stewart, M.; **2007**. Decision support frameworks and metrics for sustainable development of minerals and metals, *Clean Technology and Environmental Policy*, 9: 133-145.

Petrie, J.; **2007**. New models of sustainability for the resources sector: A focus on minerals and metals, *IChemE Journal*, 85: (B1): 88-98.

Resolute Gold Mining; **2007**. *Resolute Operations Overview: Golden Pride* [online], Available from: http://www.resolute-ltd.com.au/op_goldenpride.html, Accessed 04/03/2008.

Resolute Tanzania Limited; **1999**. *Golden Pride Project Tailings Pump and Pipeline System: Small Scale Pipe Loop Test: Report No. GPM-223.R03*, Patterson & Cooke Consulting Engineers, Cape Town, South Africa, pp 2.

Riekkola-Vanhanen, M., **1999**. *Finnish expert report on best available techniques in copper production and by-production of precious metals: Report*, Finnish Environment Institute, Helsinki, Finland.

Robèrt, K-H.; **2000**. Tools and concepts for sustainable development, how do they relate to a general framework for sustainable development, and to each other?, *Journal of Cleaner Production*, 8 (3): 243-254.

Robèrt, K-H., Schmidt-Bleek, B., de Larderel, J., Basile, G., Jansen, J., Kuehr, R., Thomas, P., Suzuki, M. and Wackernagel, M.; **2002**. Strategic sustainable development –

selection, design and synergies of applied tools, *Journal of Cleaner Production*, 10: 197-214.

Rosenhead, J.; **1989**. Rational Analysis for a Problematic World. Problem Structuring Methods for Complexity, Uncertainty and Conflict. Chichester, West Sussex: John Wiley & Sons, pp. 2-11.

Rüdenauer, I., Gensch, C-O., Grießhammer, R. and Bunke, D.; **2005**. Integrated Environmental and Economic Assessment of Products and Processes: A Method for Eco-efficiency Analysis, *Journal of Industrial Ecology*, 9: (4): 102-116.

Saling, P., Kicherer, A., Dittrich-Krämer, B., Wittlinger, R., Zombik, W., Schmidt, I., Schrott, W. and Schmidt, S.; **2002**. Eco-efficiency analysis by BASF: the method, *International Journal of Life Cycle Assessment*, 7 (4): 203-218.

Sandgren, P., Berglind, B. and Modigh, S.; **2004**. *Metso Minerals Handbook: Basics in Minerals Processing*, Helsinki: Metso Minerals (AB), pp 1-2.

Scholz, R. W. and Wiek, A.; **2005**. Operational Eco-efficiency: Comparing Firms' Environmental Investments in Different Domains of Operation, 9: (4): 155-170.

Scott, J.W.; **2002**. *Design Criteria: The Formal Basis for Design*, **Mineral Processing Plant Design, Practice and Control Symposium 2002**, Vancouver, Canada; Littleton, CO: SME, pp 3-21.

Sebitosi, A.B. and Pillay, P.; **2008**. Grappling with a half-hearted policy: The case of renewable energy and the environment in South Africa, *Energy Policy*, 36 (7): 2513-2516.

Seppälä, J., Melanen, M., Mäenpää, I., Koskela, S., Tenhunen, J. and Hiltunen, M-J.; **2005**. How Can the Eco-efficiency of a Region be Measured and Monitored?, *Journal of Industrial Ecology*, 9: (4): 117-130.

Shukla, P. and Deb, K.; **2007**. On finding multiple Pareto-optimal solutions using classical and evolutionary generation methods, *European Journal of Operational Research*, 181: 1630–1652.

Sigma; **2007**. *The Sigma Guidelines: Putting sustainable development into practice - A guide for organisations: Report*, London, UK, pp 4.

Singleton, W.; **1985**. Man-machine systems. In J. Beishon and G. Peters (Ed.), *Systems Behaviour* pp. 119-130, London: Harper & Row, pp 22.

Stewart, M. and Petrie, J.; **2006**. A process systems approach to life cycle inventories for minerals: South African and Australian case studies, *Journal of Cleaner Production*, 14: 1042-1056.

Stewart, M., Basson, L. and Petrie, J.; **2003**. Evolutionary design for environment in minerals processing, *IChemE Journal*, 81 (B): 341-351.

Stewart, M.; **1999**. *Environmental life cycle considerations for design-related decision-making in minerals processing*, PhD Thesis, **University of Cape Town**, Cape Town, South Africa, pp 2-1 – 2-32, 3-1, 4-1, 5-1.

STOWA; **2006**. *Selected Dewatering Technologies – Fact Sheets: Report*, Stitching Toegepast Onderzoek, Amsterdam, the Netherlands, pp 1-12.

Swinkels, P., van der Weidjen, R., Ajah, A., Arifin, Y., Loe, H., Manik, M. and Reuter, M.; **2004**. Conceptual process design as a prerequisite for solving environmental problems: A case study of molybdenum removal and recovery from wastewater, *Minerals Engineering*, 17: 205-215.

Tahara, K., Mizuguchi, T., Mizuno, T. and Inaba, A.; **2006**. *Eco-Efficiency indicator for evaluation of companies' environmental performances*, **2nd International *Conference* on Quantified Eco-Efficiency Analysis for Sustainability 2006**, Egmond aan Zee, the Netherlands; Leiden: Leiden University, pp. 1-12.

Turton, R., Bailie, C., Wallace, B. and Joseph, A.; **1998**. *Analysis, synthesis and design of chemical processes*, 2nd ed., Upper Saddle River, NJ: Prentice Hall, pp 51-88, 221-252.

UNEP; **1995**. *Cleaner production worldwide*, vol. 2., Paris: United Nations Environment Programme, pp 1-69.

UNESCO; **2003**. *Water for People, Water for Life: Report, United Nations World Water Development*, Paris, France, pp 5-23.

van Berkel, R. and Narayanaswamy, V.; **2005**. *Eco-efficiency for design and operation of mineral processing plants*, **CHEMECA 2005**, Brisbane, Australia; Melbourne, VA: AusIMM, pp 1-5.

van Berkel, R.; **2007a**. Eco-efficiency in the Australian minerals sector, *Journal of Cleaner Production*, 15: 772-781.

van Berkel, R.; **2007b**. Eco-efficiency in primary metals production: Context, perspectives and methods, *Resources, Conservation and Recycling*, 51: 511-540.

van Zyl, D., Sasson, M., Digby, C., Fleury, A.-M. and Kyeyune, S.; **2002**. *Mining for the Future: Report No. 68*, Minerals, Metals and Sustainable Development Project, London, United Kingdom, pp C-1 – C-23.

Vergidis, K., Tiwari, A., Majeed, B. and Roy, R.; **2007**. Optimisation of business process designs: An algorithmic approach with multiple objectives, *International Journal of Production Economics*, 109: 105-121.

von Blottnitz, H.; **2008**. *Personal communication*.

Wacker, J.G.; **1998**. A definition of theory: research guidelines for different theory-building research methods in operations management, *Journal of Operations Management*, 16: 361-385.

Wakeman, R.; **2007**. The influence of particle properties on filtration, *Separation and Purification Technology*, 58: 234-241.

WBCSD; **2000**. *Eco-efficiency: Creating more value with less impact: Report*, Geneva, Switzerland, pp 1-36.

WCED; **1987**. *Our Common Future: Report*, Geneva, Switzerland, pp 2.

Weidema, B.P.; **1998**. Application Typologies for Life Cycle Assessment, *International Journal of Life Cycle Assessment*, 3 (4): 237-240.

While, L., Barone, L., Hingston, P., Huband, S., Tuppurainen, D. and Bearman, R.; **2004**. A multi-objective evolutionary algorithm approach for crusher optimisation and flowsheet design, *Minerals Engineering*, 17: 1063-1074.

Winkler, H., **2006**. Energy policies for sustainable development in South Africa's residential and electricity sectors: Implications for mitigating climate change. PhD Thesis, **University of Cape Town**, Cape Town, South Africa, pp 40.

Winpenny, J.T., **2000**. Managing Water Scarcity for Water Security, United Nations Food & Agricultural Organisation, [online] Available from: http://www.fao.org/ag/agl/aglw/webpub/scarcit1.htm, Accessed on 16/08/2008.

Woodside, A.G. and Wilson, E.J.; **2003**. Case study methods for theory building, *Journal of Business and Industrial Marketing*, 18 (6/7): 493-508.

Wrisberg, N. and Triebswetter, U.; **1999**. Draft Input to Chainet Guidebook: Demands for Environmental Information. In *Centre of Environmental Science* (ed.), Leiden: Leiden University, pp 11-20.

Wu, K., Yu, C., Luyben, W. and Skogestad, S.; **2002**. Reactor/separator processes with recycles - 2. Design for composition control, *Computers and Chemical Engineering*, 27: 401-421.

Yang, Y., Shi, L.; **2000**; Integrating environmental impact minimization into conceptual chemical process design – a process systems engineering review, *Computers and Chemical Engineering*, 24: 1409-1419.

Yin, R.; **1994**. *Case study research: design and methods*, 2nd ed., Beverley Hills, CA: Sage Publishing, pp 6-12.

Zhelev, T.; **2007**. The conceptual design approach – A process integration approach on the move, *Resources, Conservation and Recycling*, 50 (2): 143-157.

APPENDICES

Table of Contents (Appendices)

APPENDIX A

Techno-environmental Performance Assessment

The assumptions, raw input data and computational results for the environmental performance assessments presented in Chapter 4 and Chapter 5 of this thesis has been included in this section. Technical process information and performance data on which the assessments were based are also included for completion.

A.1 Case Study 1

A.1.1 Assumptions and raw data

A brief set of technical process descriptions characterizing the unit technologies with which the three copper processing routes were developed has been included in Table A.1 below.

Table A.1: Brief descriptions of technology options constituting the copper processing routes

Name	Description
Reverbaratory furnace	Smelter for dried concentrate Produces molten matte (50-60wt% copper) and molten slag (~1wt% copper). Energy input: electricity, fuel oil
Blister flash smelter	Cu ore concentrate in oxygen-enriched air by oxidation of sulphur and iron that is contained within the copper mineral. Heat is required to melt the concentrate is provided by the exothermic oxidation reactions and by burning small amounts of fossil fuel.
Heap leach	Cu leaching in dilute H_2SO_4 - final solution of 1.25 - 15 g/L
Solvent extraction	Countercurrent exchange of organic and aqueous solutions Resultant concentration = 36 - 58 g/L
Electrorefining	High grade Cu dissolved into an electrolyte from which copper cathode metal is subsequently electroplated
Electrowinning	Similar to electrolytic refining, but without the metal dissolution stage

Model assumptions used to generate material balances for the reverbaratory smelting, flash smelting and HL-SX-EW processes have been shown in Table A.2, Table A.3 and Table A.4, respectively.

Table A.2: Full set of model assumptions for the reverbaratory smelting process (from Giurco, 2005)

Underground		Reference
MineUG_Elec_Rate_kWh_per_t ore	95	(MIM, 1999)
MineUG_Water_Rate_t_per_t ore	1.16	(MIM, 1999)
MineUG_Diesel_Rate_t_per_t ore	0	not known
Open Cut		
MineOpen_Elec_Rate_kWh_per_t ore	20	(MIM, 1999)
MineOpen_Water_Rate_t_per_t ore	0.58	(MIM, 1999)
MineOpen_Diesel_Rate_t_per_t ore	0.002	(Norgate and Rankin, 2000)
Mine_CO2_Rate_t_per_t_Diesel	3.64	(PRè, 2000)
Opencut_Rate_percent	95%	(Biswas and Davenport, 1994)
Concentrate-Ore Properties		
Ore_Cu_Rate_percent	0.5%	(Ayres et al., 2001)
Ore_Fe_Rate_percent	calculated	
Ore_S_Rate_percent	calculated	
Ore_SiO2_Rate_percent	calculated	
Ore_Contaminants_Rate_percent	calculated	
Ore_Balance(Gangue)_Rate_percent	calculated	
Conc_Cu_Recovery_percent	90%	(Biswas and Davenport, 1994)
Tail_Fe_Cu_Ratio	0.88	Assumes all Cu not recovered which ends up tailings is associated with CuFeS2. So M(Cu)/M(Fe)=1.14, hence ratio Fe:Cu = 1/1.14 = 0.88
Tail_S_Cu_Ratio	1.00	m(S)/m(Cu) = 2*M(S)/M(Cu) = 2*32/64=1
Tail_SiO2_Cu_Ratio	2.00	Estimate
Tail_Contaminants_Cu_Ratio	0.10	Estimate
Conc_Cu_Rate_percent	30%	(Biswas and Davenport, 1994)
Conc_Fe_Rate_percent	27%	(Biswas and Davenport, 1994)
Conc_S_Rate_percent	30%	(Biswas and Davenport, 1994)
Conc_SiO2_Rate_percent	7%	(Biswas and Davenport, 1994)
Conc_Contaminants_Rate_percent	1%	(Biswas and Davenport, 1994)
Conc_Balance(Gangue)_Rate_percent	5%	(Biswas and Davenport, 1994)
Smelting		
Smelt_Oil_Rate_t_per_t_conc	0.141	(Biswas and Davenport, 1994)
Smelt_Elec_Rate_kWh_per_t conc	120	(Riekkola-Vanhanen, 1999)
Smelt_SiO2_Rate_t_per_t_conc	0.1	(Biswas and Davenport, 1994)
Smelt_O2_Rate_t_per_t_conc	0.1	(Biswas and Davenport, 1994)
Smelt_Slag_Recycle_t_per_t_conc	0.05	(Biswas and Davenport, 1994)
Smelt_Slag_Cu_Rate_percent	0.9%	(Biswas and Davenport, 1994)
Smelt_Matte_Cu_Rate_percent	45%	(Biswas and Davenport, 1994)
Smelt_CO2_Rate_t_per_t_oil	3.82	(PRè, 2000)
Smelt_SO2_Rate_t_per_t Cu_in_conc	2	from conc. composition
Converting		
Convert_O2_Rate_t_per_t conc	0.224	(Biswas and Davenport, 1994)
Convert_SO2_Rate_t	0	assume all in smelter
Convert_Slag_Cu_Rate_percent	2.50%	(Biswas and Davenport, 1994)
Blister_Cu_Rate_percent	0.997	(Biswas and Davenport, 1994)

Electro-refining		
ERefine_Elec_Rate_kWh_t Cu	400	(MIM, 1999)
ERefine_Water_Rate_t_t Cu	1	(MIM, 1999)
ERefine_Effluent_Rate_t_t Cu	0.128	(MIM, 1999)
ERefine_Metals_Rate_t_t Cu	0.00000565	(MIM, 1999)
Gas Treatment		
GasTreat_O2_Rate_t_t SO2	*0.2325*	*By stoichiometry 2SO2+O2-->2SO3 [0.5*32/64*percent to acid]*
GasTreat_Water_Rate_t_t SO2	*0.2615625*	*By stoichiometry SO3+H2O-->H2SO4 [18/64*percent to acid]*
GasTreat_SO2_to_acid_Rate_percent	5%	(Biswas and Davenport, 1994; Riekkola-Vanhanen, 1999)
GasTreat_Elec_Rate_kWh_t_conc	95	(Riekkola-Vanhanen, 1999)

Table A.3: Full set of model assumptions for the flash smelting process (from Giurco, 2005)

Underground		Reference
MineUG_Elec_Rate_kWh_per_t ore	95	(MIM, 1999)
MineUG_Water_Rate_t_per_t ore	1.16	(MIM, 1999)
MineUG_Diesel_Rate_t_per_t ore	0	not known
Open Cut		
MineOpen_Elec_Rate_kWh_per_t ore	20	(MIM, 1999)
MineOpen_Water_Rate_t_per_t ore	0.58	(MIM, 1999)
MineOpen_Diesel_Rate_t_per_t ore	0.002	(Norgate and Rankin, 2000)
Mine_CO2_Rate_t_per_t_Diesel	3.64	(PRè, 2000)
Opencut_Rate_percent	95%	(Biswas and Davenport, 1994)
Concentrate-Ore Properties		
Ore_Cu_Rate_percent	0.5%	(Ayres et al., 2001)
Ore_Fe_Rate_percent	calculated	
Ore_S_Rate_percent	calculated	
Ore_SiO2_Rate_percent	calculated	
Ore_Contaminants_Rate_percent	calculated	
Ore_Balance(Gangue)_Rate_percent	calculated	
Conc_Cu_Recovery_percent	90%	(Biswas and Davenport, 1994)

Tail_Fe_Cu_Ratio	0.88	Assumes all Cu not recovered which ends up in tailings is associated with CuFeS2. So M(Cu)/M(Fe)=1.14, hence ratio Fe:Cu = 1/1.14 = 0.88
Tail_S_Cu_Ratio	1.00	m(S)/m(Cu) = 2*M(S)/M(Cu) = 2*32/64=1
Tail_SiO2_Cu_Ratio	2.00	Estimate
Tail_Contaminants_Cu_Ratio	0.10	Estimate
Conc_Cu_Rate_percent	30%	(Biswas and Davenport, 1994)
Conc_Fe_Rate_percent	27%	(Biswas and Davenport, 1994)
Conc_S_Rate_percent	30%	(Biswas and Davenport, 1994)
Conc_SiO2_Rate_percent	7%	(Biswas and Davenport, 1994)
Conc_Contaminants_Rate_percent	1%	(Biswas and Davenport, 1994)
Conc_Balance(Gangue)_Rate_percent	5%	(Biswas and Davenport, 1994)
Smelting		
Smelt_Oil_Rate_t_per_t_conc	0.044	(Riekkola-Vanhanen, 1999)
Smelt_Elec_Rate_kWh_per_t conc	120	(Riekkola-Vanhanen, 1999)
Smelt_SiO2_Rate_t_per_t_conc	0.1	(Biswas and Davenport, 1994)
Smelt_O2_Rate_t_per_t_conc	0.2	(Biswas and Davenport, 1994)
Smelt_Slag_Recycle_t_per_t_conc	0.05	(Biswas and Davenport, 1994)
Smelt_Slag_Cu_Rate_percent	1.5%	(Biswas and Davenport, 1994)
Smelt_Matte_Cu_Rate_percent	62%	(Biswas and Davenport, 1994)
Smelt_CO2_Rate_t_per_t_oil	3.82	(PRè, 2000)
Smelt_SO2_Rate_t_per_t Cu_in_conc	2	from conc. composition
Converting		
Convert_O2_Rate_t_per_t conc	0.127	(Biswas and Davenport, 1994)
Convert_SO2_Rate_t	0	assume all in smelter
Convert_Slag_Cu_Rate_percent	2.50%	(Biswas and Davenport, 1994)
Blister_Cu_Rate_percent	0.997	(Biswas and Davenport, 1994)
Electro-refining		
ERefine_Elec_Rate_kWh_t Cu	400	(MIM, 1999)
ERefine_Water_Rate_t_t Cu	1	(MIM, 1999)
ERefine_Effluent_Rate_t_t Cu	0.128	(MIM, 1999)
ERefine_Metals_Rate_t_t Cu	0.00000565	(MIM, 1999)
Electric Slag Cleaning Furnace		
SlagClean_Elec_Rate_kWh_t_Cu	40	(Biswas and Davenport, 1994)
SlagClean_Slag_Cu_Rate_percent	0.50%	(MIM, 1999)
SlagClean_Slag_Fe_Rate_percent	36.7%	(MIM, 1999)
SlagClean_Slag_S_Rate_percent	0.5%	(MIM, 1999)
SlagClean_Slag_SiO2_Rate_percent	36%	(MIM, 1999)
SlagClean_Slag_Contaminants_Rate_percent	2%	Estimate
SlagClean_Slag_Balance_Rate_percent	24%	By difference
Gas Treatment		
GasTreat_O2_Rate_t_t SO2	*0.2325*	*By stoichiometry 2SO2+O2-->2SO3 [0.5*32/64*percent to acid]*
GasTreat_Water_Rate_t_t SO2	*0.2615625*	*By stoichiometry SO3+H2O-->H2SO4 (18/64*percent to acid)*
GasTreat_SO2_to_acid_Rate_percent	93%	(Riekkola-Vanhanen, 1999)
GasTreat_Elec_Rate_kWh_t_conc	120	(Riekkola-Vanhanen, 1999)

Table A.4: Full set of model assumptions for the HL-SX-EW process (from Giurco, 2005)

Mining		Reference
Mining_Elec_Rate_kWh_per_t_ore	13	(Norgate and Rankin, 2000)
Mining_Diesel_Rate_t_t_ore	0.002	(Norgate and Rankin, 2000)
Crushing		
Crushing_Elec_Rate_kWh_per_t_ore	2	(Norgate and Rankin, 2000)
Ore		As for flash/reverb
Ore_Cu_Rate_percent	0.5%	As for flash/reverb
EW_RefinedCu_Cu_Rate_percent	100%	Product Quality
Liq_Cu_Recovery_per_t_ore	65%	(Biswas and Davenport, 1994; Jenkins et al., 1999; Norgate and Rankin, 2000)
Liq_Fe_Recovery_per_t_ore	80%	
Liq_S_Recovery_per_t_ore	65%	
Liq_SiO2_Recovery_per_t_ore	0%	
Liq_Contaminants_Recovery_per_t_ore	10%	
Liq_Balance(Gangue)_Recovery_per_t_ore	0%	
Solvent Extraction		
SX_Elec_Rate_kWh_per_t Cu	2500	(Norgate and Rankin, 2000)
SX_Steam_Rate_t_per_t Cu	0.23	(Norgate and Rankin, 2000)
SX_H2SO4_Rate_t_per_t_Cu	1.7	(Biswas and Davenport, 1994)
Electrowinning		
EW_Elec_Rate_Kwh_per_t Cu	2000	(Biswas and Davenport, 1994)

The above models were extended to include indirect global warming and resource depletion environmental impacts, as explained in Chapter 4. The key assumptions underpinning this analysis have been shown in Table A.5 below.

Table A.5: Assumptions relating to the consideration of off-site resource depletion and global warming impacts

A. Crude oil-based liquid fuels (Diesel and heavy fuel oil)

- 100% Iranian sweet brent crude oil, transported via sea transport (+/- 7,000 tkm) to South Africa
- Crude oil production (including drilling and pre-processing)
- Pipeline transport from the oil rig to onshore facilities
- Tanker transport to South African ports
- Liquid fuels production at South African crude oil refineries (Calref, Enref, Natref and Sapref)
- Local pipeline or road transport of diesel or fuel oil to site is not included in the analysis
- Source: Goedkoop *et. al.* (2008)

B. Coal-based power generation

- 100% hard coal fuel mix (estimated SA hard coal fuel input approx. 95% - ESKOM, 2006)
- An emission factor of **1.01 kg/kWh** can be applied (ESKOM, 2006)

The design was carried out in a Microsoft Excel™ environment. The detailed process models for each alternative are available as Appendix E in electronic format on the CD-ROM accompanying this thesis.

A.1.2 Results data

Using the assumptions presented the in the preceding section, an overall material balance was therefore computed for each of the three copper processing routes. Simplified energy balance calculations were used to estimate the energy requirements for each process. Table A.6, Table A.7 and Table A.8 below show the material balance and energy consumption associated with each copper processing route investigated.

Table A.6: Overall input-output table for the reverbaratory smelting process in tonnes per annum

COMPONENTS	INPUT	OUTPUT	WASTE
Cu	165,333	145,338	16,533
Fe	140,017	-	14,467
S	156,033	-	16,533
SiO2	65,617	-	33,067
Contaminant/Precious Metals	6,303	-	1,654
Gangue	24,666,107	-	24,621,157
O2	155,355	-	-
CO2 direct	-	-	432,262
CO2 indirect	-	-	866,069
SO2	-	-	352,915
H2SO4	-	381,357	-
Water	16,227,590	-	16,095,827
Fuel Oil	65,565	-	-
Diesel	45,359		
Electricity (kWh)	851,093,915	-	-

Ore feed rate	**25,199,411**	**tpa**
Cu production rate	**145,338**	**tpa**
Tailings discharge rate	**24,703,411**	**tpa**

Table A.7: Overall input-output table for the flash smelting process in tonnes per annum

COMPONENTS	INPUT	OUTPUT	WASTE
Cu	165,333	144,690	17,229
Fe	148,387	-	65,541
S	165,333	-	17,229
SiO2	67,787	-	83,445
Contaminant/Precious Metals	6,613	-	4,872
Gangue	23,065,594	-	23,074,334
O2	231,384	-	-
CO2 direct	-	-	246,717
CO2 indirect	-	-	759,912
SO2	-	-	20,832
H2SO4	-	444,633	-
Water	14,606,531	-	14,402,576
Fuel Oil	21,824	-	-
Diesel	44,878	-	-
Electricity (kWh)	745,457,276	-	-

Ore feed rate	**23,619,048**	**tpa**
Cu production rate	**144,690**	**tpa**
Tailings discharge rate	**23,262,651**	**tpa**

Table A.8: Overall input-output table for the HL-SX-EW smelting process in tonnes per annum

COMPONENTS	INPUT	PRODUCT	WASTE
Cu	247,863	145,000	102,863
Fe	222,892	-	62,410
S	249,337	-	103,475
SiO2	102,001	-	102,001
Contaminant/Precious Metals	11,333	-	10,313
Gangue	36,947,147	-	36,947,147
O2	-	-	-
CO2 direct	-	-	275,026
CO2 indirect	-	-	1,713,465
SO2	-	-	-
H2SO4	-	-	-
Water	24,102,536	-	24,105,871
Fuel Oil	-	-	-
Diesel	75,557	-	-
Electricity (kWh)	1,440,176,988	-	-

Ore feed rate	**37,780,574**	**tpa**
Cu production rate	**145,000**	**tpa**
Waste rock	**37,328,210**	**tpa**

Greenhouse gas emissions and **water consumption** figures were therefore computed directly from the material balance data shown above. However, to determine the **eco-toxicity** and **resource depletion** potentials for each design alternatives, additional ore characterisation calculations needed to be undertaken, based on average metallic concentrations of major and moderately abundant metals in a poryphry-type copper sulphide ore. These computations are presented in Table A.9 – Table A.16 below.

1) REVERBARATORY SMELTING

<u>Assumption Set</u>: Massive Cu sulphide ore deposit
Only sulphide ore-forming elements are considered at conceptual design
Silver (Ag) and gold (Au) are recovered with copper (Cu) during electrorefining (Biswas and Davenport, 1994)
-> Assume 70% recovery (range 70%-100%; Ayres *e.t al.*, 2002)

Table A.9: Aquatic eco-toxicity data for the reverbaratory smelting process

Component	Tailings concentrations		Discharge rate (tpa)		Relative discharge (tonne/tonne Cu)		Aquatic Ecotoxicity kg/ kg 1.4-DB-equivalents
	Min	Max	Min	Max	Min	Max	
Major elements (mass %)							
Cu	0.5	1	123517	247034	0.850	1.700	1.16E+03
Fe	1	10	247034	2470341	1.700	16.997	0.00E+00
S	2	11	494068	2717375	3.399	18.697	
Moderately abundant elements (ppm)							
Zn	150	500	3706	12352	0.025	0.085	7.21E+03
Pb	5	100	124	2470	0.001	0.017	1.11E+03
As	2	550	49	13587	0.000	0.093	1.19E+05
Mo	4	450	99	11117	0.001	0.076	2.62E+06
Bi	0.2	60	5	1482	0.0000	0.010	
Sb	0.2	60	5	1482	0.0000	0.010	1.23E+03
Cd	1	50	25	1235	0.0002	0.008	2.20E+05
Ni	1	50	25	1235	0.0002	0.008	2.25E+06
Se	1	50	25	1235	0.0002	0.008	2.53E+07

Table A.10: Resource depletion data for the reverbaratory smelting process

Component	Ore concentrations		Feed rate (tpa)		Relative feed (tonne/tonne Cu)		Resource Depletion
	Min	Max	Min	Max	Min	Max	kg Sb-equivalents
Major elements (mass %)							

Cu	0.5	1	125997	251994	0.867	1.734	2.20E-05
Fe	1	10	251994	2519941	1.734	17.339	8.43E-08
S	2	11	503988	2771935	3.468	19.072	
Moderately abundant elements (ppm)							
Zn	150	1600	3780	40319	0.026	0.277	9.92E-04
Pb	30	300	756	7560	0.005	0.052	1.35E-02
As	5	1800	126	45359	0.001	0.312	9.17E-03
Mo	15	1500	378	37799	0.003	0.260	3.17E-02
Bi	2	200	50	5040	0.000	0.035	0.0731
Sb	2	200	50	5040	0.000	0.035	3.30E-02
Cd	2	200	50	5040	0.000	0.035	3.30E-01
Ni	8	150	202	3780	0.001	0.026	1.80E-04
Se	10	100	252	2520	0.002	0.017	4.75E-01

Resource depletion from fossil fuels	**tonne Sb-eq.**	**tonne Sb-eq./tonne Cu**
Diesel	1066	0.007
Fuel oil	1525	0.010

Table A.11: Byproduct value recovery data for the reverbaratory smelting process

Element	**Ore concentration range (ppm)**	**Source**	**Assumed concentration (ppm)**	**Recovery (%)**	**Final product (tpa)**
Ag	1-70	Broadhurst (2007a)	40	70%	705.6
Au	0.4-4	Broadhurst (2007a)	1	70%	17.6

2) FLASH SMELTING

<u>Assumption Set</u>: Massive Cu sulphide ore deposit
Only sulphide ore-forming elements are considered at conceptual design
Silver (Ag) and gold (Au) are recovered with copper (Cu) during electrorefining (Biswas and Davenport, 1994)
-> Assume 70% recovery (range 70%-100%; Ayres *et. al.*, 2002)

Table A.12: Aquatic eco-toxicity data for the flash smelting process

Component	Tailings concentrations		Discharge rate (tpa)		Relative discharge (tonne/tonne Cu)		Aquatic Ecotoxicity kg/ kg 1.4-DB-equivalents
	Min	Max	Min	Max	Min	Max	
Major elements (mass %)							
Cu	0.5	1	116313	232627	0.804	1.608	1.16E+03
Fe	1	10	232627	2326265	1.608	16.078	0.00E+00
S	2	11	465253	2558892	3.216	17.685	
Moderately abundant elements (ppm)							
Zn	150	500	3489	11631	0.024	0.080	7.21E+03
Pb	5	100	116	2326	0.001	0.016	1.11E+03
As	2	550	47	12794	0.000	0.088	1.19E+05
Mo	4	450	93	10468	0.001	0.072	2.62E+06
Bi	0.2	60	5	1396	0.000	0.010	
Sb	0.2	60	5	1396	0.000	0.010	1.23E+03
Cd	1	50	23	1163	0.000	0.008	2.20E+05
Ni	1	50	23	1163	0.000	0.008	2.25E+06
Se	1	50	23	1163	0.000	0.008	2.53E+07

Table A.13: Resource depletion data for the flash smelting process

Component	Ore concentrations		Feed rate (tpa)		Relative feed (tonne/tonne Cu)		Resource Depletion
	Min	Max	Min	Max	Min	Max	kg Sb-equivalents
Major elements (mass %)							
Cu	0.5	1	118095	236190	0.816	1.632	2.20E-05
Fe	1	10	236190	2361905	1.632	16.324	8.43E-08
S	2	11	472381	2598095	3.265	17.956	
Moderately abundant elements (ppm)							
Zn	150	1600	3543	37790	0.024	0.261	9.92E-04
Pb	30	300	709	7086	0.005	0.049	1.35E-02
As	5	1800	118	42514	0.001	0.294	9.17E-03
Mo	15	1500	354	35429	0.002	0.245	3.17E-02
Bi	2	200	47	4724	0.000	0.033	0.0731
Sb	2	200	47	4724	0.000	0.033	3.30E-02
Cd	2	200	47	4724	0.000	0.033	3.30E-01
Ni	8	150	189	3543	0.001	0.024	1.80E-04
Se	10	100	236	2362	0.002	0.016	4.75E-01

Resource depletion from fossil fuels	**tonne Sb-eq.**	**tonne Sb-eq./tonne Cu**
Diesel	1054	0.007
Fuel oil	508	0.004

Table A.14: Byproduct value recovery data for the flash smelting process

Element	**Ore concentration range (ppm)**	**Source**	**Assumed concentration (ppm)**	**Recovery (%)**	**Final product (tpa)**
Ag	1-70	Broadhurst (2007a)	40	70%	661.3
Au	0.4-4	Broadhurst (2007a)	1	70%	16.5

3) HEAP LEACH/SOLVENT EXTRACTION/ELECTROWINNING

Assumption Set:

Massive Cu sulphide ore deposit
Only sulphide ore-forming elements are considered at conceptual design
No recovery of Au and Ag by-products occurs

Table A.15: Aquatic eco-toxicity data for the HL-SX-EW process

Component	Tailings concentrations		Discharge rate (tpa)		Relative discharge (tonne/tonne Cu)		Aquatic Ecotoxicity kg/ kg 1.4-DB-equivalents
	Min	Max	Min	Max	Min	Max	
Major elements (mass %)							
Cu	0.5	1	186641	373282	1.287	2.574	1.16E+03
Fe	1	10	373282	3732821	2.574	25.744	0.00E+00
S	2	11	746564	4106103	5.149	28.318	0
Moderately abundant elements (ppm)							
Zn	150	1600	5599	59725	0.039	0.412	7.21E+03
Pb	30	300	1120	11198	0.008	0.077	1.11E+03
As	5	1800	187	67191	0.001	0.463	1.19E+05
Mo	15	1500	560	55992	0.004	0.386	2.62E+06
Bi	2	200	75	7466	0.001	0.051	
Sb	2	200	75	7466	0.001	0.051	1.23E+03
Cd	2	200	75	7466	0.001	0.051	2.20E+05
Ni	8	150	299	5599	0.002	0.039	2.25E+06
Se	10	100	373	3733	0.003	0.026	2.53E+07

Table A.16: Resource depletion data for the HL-SX-EW process

Component	Ore concentrations		Feed rate (tpa)		Relative feed (tonne/tonne Cu)		Resource Depletion
	Min	Max	Min	Max	Min	Max	kg Sb-equivalents
Major elements (mass %)							
Cu	0.5	1	188903	377806	1.303	2.606	2.20E-05
Fe	1	10	377806	3778057	2.606	26.056	8.43E-08
S	2	11	755611	4155863	5.211	28.661	
Moderately abundant elements (ppm)							
Zn	150	1600	5667	60449	0.039	0.417	9.92E-04
Pb	30	300	1133	11334	0.008	0.078	1.35E-02
As	5	1800	189	68005	0.001	0.469	9.17E-03
Mo	15	1500	567	56671	0.004	0.391	3.17E-02
Bi	2	200	76	7556	0.001	0.052	0.0731
Sb	2	200	76	7556	0.001	0.052	3.30E-02
Cd	2	200	76	7556	0.001	0.052	3.30E-01
Ni	8	150	302	5667	0.002	0.039	1.80E-04
Se	10	100	378	3778	0.003	0.026	4.75E-01

Resource depletion from fossil fuels	**tonne Sb-eq.**	**tonne Sb-eq./tonne Cu**
Diesel	1775	0.012
Fuel oil	0	0

A.2 Case Study 2

A.2.1 Assumptions and raw data

Previous analytical work (Mwakyusa, 2007) has been carried out to characterise the solids discharged from the GPP process plant to the dewatering circuit prior to disposal in the tailings dam. The tailings feed particle size distribution (PSD) was used as input data to calibrate thickener and hydrocyclone performance in the process models generated. The feed PSD has been included in Table A.17 below.

Table A.17: Dewatering circuit feed particle size distribution data (Mwakyusa, 2007)

Size (□m)	Weight (g)	Mass fraction [mi]	Screen size mean [di] (m)	[di/mi]	Cum% Retained	Cum% Passing
300	0.23	0.00046	0.0003625	0.788	0.05	99.95
212	6.43	0.01286	0.0003185	0.025	1.33	98.67
150	29.86	0.05972	0.0002875	0.005	7.3	92.7
106	35.25	0.0705	0.0002655	0.004	14.35	85.65
75	49.14	0.09828	0.00025	0.003	24.18	75.82
53	26.81	0.05362	0.000239	0.004	29.54	70.46
38	67.92	0.13584	0.0002315	0.002	43.13	56.87
25	124.12	0.24824	0.000225	0.001	67.95	32.05
0	160.24	0.32048	0.0002125	0.001	100	0
TOTAL	500	1		0.832		

The above data were used together with technical process criteria for each of the dewatering units used to construct the dewatering circuit design alternatives, shown in Table A.18.

Table A.18: Key technical assumptions for the dewatering circuit (from Mwakyusa, 2007)

Assumptions	Value
Overall	
Plant solids throughput (t/hr)	400
Plant slurry throughput (t/hr)	800
Estimated tailings dam area (m^2)	700,000
Thickening	
Thickener solids recovery to underflow	100%
Lamella solids recovery to underflow	100%
Target conventional thickener underflow % solids	60%
Target high-rate thickener underflow % solids	70%
Hydrocyclone separation	
Dc (cm)	38.1
Di (cm)	10.5
Inlet Area/Cross-sectional Area	7.6%
Overerflow diameter [Do] (cm)	13.0
Underflow diameter [Du] (cm)	12.5

Height [h] (cm)	91.0
Volumetric flowrate solids (m^3/hr)	15.9
Volumetric flowrate water (m^3/hr)	46.7
Feed volume % solids	25
S (g/cm^3)	2.9
P (kPa)	120.0
% solids Overflow	32
d50 (μm)	22.710
Qmax (m^3/hr)	149.475
Qtheoretical (m^3/hr)	74.141
Target filter underflow % solids	79%
Filtration	
Filter solids recovery to underflow	100%
Acceleration by gravity (ms^{-2})	9.8
Density solids (g/cm^3)	2.8
Density liquid (g/cm^3)	1.0
Dynamic viscosity liquid (Nms^{-2})	0.001
Kozeny coefficient	5.0
Pressure drop (kPa)	150
Viscosity (N.s/m^2)	0.001
Liquid density (kg/m^3)	1000
Cake thickness (m)	0.002
Average particle diameter (m)	0.00012
Porosity	0.4
Kozeny constant	5
Filtration area (m^2)	0.01
Specific area (m^2/m^3)	50000
Specific resistance [r] (m^{-2})	7.1E-07

An overall water balance for the GPP case study was developed by Mwakyusa (2007). This water balance set the computational context for a detailed water balance for the dewatering circuit, performed in this thesis. The overall water balance has been shown in Figure A.1 below.

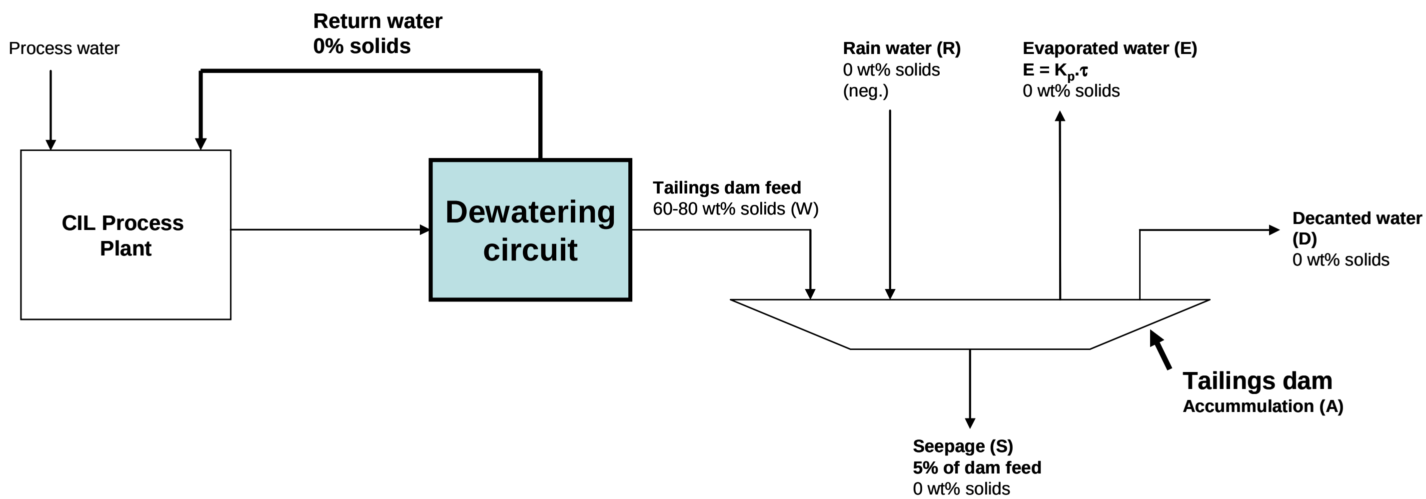

Figure A.1: Overview of GPP water balance (Mwakyusa, 2007)

Therefore, the water balance over the tailings dam can be written as:

$$(F + R) - (E + D + S) = A$$

Equation A.1: Mathemeatical representation of overall GPP water balance

Where K_p = evaporation coefficient, dependent on the local surface evaporation rate

τ = average daily evaporation rate (measured from a local rain gauge in mm)

A.2.2 Results data

The material balance for each of the dewatering circuit options has been shown in Table A.19 below.

Table A.19: Detailed material balance for the six dewatering circuit alternatives

		OPTION 1	OPTION 2	OPTION 3	OPTION 4	OPTION 5	OPTION 6
NO. OF UNITS REQUIRED							
Cyclone						8	8
Conventional thickener		2		2		1	
High-rate thickener			2		2		1
Filter				14	14	10	10
TOTAL (tonnes/hr)							
	Circuit feed	**800.0**	**800.0**	**800.0**	**800.0**	**800.0**	**800.0**
	Return water	**133.3**	**228.6**	**293.7**	**293.7**	**230.8**	**279.8**
	Tailings	**666.7**	**571.4**	**506.3**	**506.3**	**569.2**	**520.2**
SLURRY (tonnes/hr)							
Cyclone	F					800.0	800.0
	O/F					394.7	275.7
	U/F					405.3	524.3
Thickener	F	800.0		800.0		394.7	
	O/F	133.3		133.3		133.3	
	U/F	666.7		666.7		261.4	
Lamella clarifier	F		800.0		800.0		275.7
	O/F		228.6		228.6		153.7
	U/F		571.4		571.4		122.0
Filter	F			666.7	571.4	405.3	524.3
	O/F			160.3	65.1	97.5	126.1
	U/F			506.3	506.3	307.8	398.2
Solids (tonnes/hr)							
Cyclone	F					400.0	400.0

	O/F					156.8	85.4
	U/F					243.2	314.6
Thickener	F	400.0		400.0		156.8	
	O/F	0.0		0.0		0.0	
	U/F	400.0		400.0		156.8	
Lamella clarifier	F		400.0		400.0		85.4
	O/F		0.0		0.0		0.0
	U/F		400.0		400.0		85.4
Filter	F			400.0	400.0	243.2	314.6
	O/F			0.0	0.0	0.0	0.0
	U/F			400.0	400.0	243.2	314.6
Water (tonnes/hr)							
Cyclone	F					400.0	400.0
	O/F					237.9	190.3
	U/F					162.1	209.7
Thickener	F	400.0		400.0		237.9	
	O/F	133.3		133.3		133.3	
	U/F	266.7		266.7		104.6	
Lamella clarifier	F		400.0		400.0		190.3
	O/F		228.6		228.6		153.7
	U/F		171.4		171.4		36.6
Filter	F			266.7	171.4	162.1	209.7
	O/F			160.3	65.1	97.5	126.1
	U/F			106.3	106.3	64.6	83.6

SOLIDS %	Circuit feed	50.0	50.0	50.0	50.0	50.0	50.0
	Tailings	**60.0**	**70.0**	**79.0**	**79.0**	**70.3**	**76.9**

The above material balance was then used to validate the overall water balance model offered by Mwakyusa (2007), shown in Table A.20 below.

Table A.20: Overall tailings water balance summary

Options	Feed Water (t/day)	Water Recovered (Dewatering) (t/day)	Water to Tailings from Feed (t/day)	Water to Tailings from Rain (t/day)	Water Loss from Evaporation (t/day)	Water Loss from Seepage (t/day)	Water Recovered (Decanting) (t/day)	Daily Water Accumulation (t/day)	**Dissipative Water Loss over discount period**
1	9600	3200	6400	0	4032.0	320.00	1156.00	892.00	**1.149.E+07**
2	9600	5486	4114	0	3584.0	205.71	1156.00	-831.43	**1.000.E+07**
3	9600	6715	2885	0	3275.4	144.27	1156.00	-1690.17	**9.028.E+06**
4	9600	7048	2552	0	3180.8	127.59	1156.00	-1912.50	**8.734.E+06**
5	9600	5539	4061	0	3571.7	203.04	1156.00	-869.97	**9.965.E+06**
6	9600	7048	2552	0	3180.8	127.59	1156.00	-1912.50	**8.734.E+06**

APPENDIX B

Economic Performance Assessment

This section details information on the economic performance assessments carried out in the case study analysis in this thesis. Key assumptions on process cost and revenue sources, as well as their variability with key process variables, have been included in this section.

B.1 Case Study 1

Revenues for each copper processing route were estimated using 2007 average metal prices at the design capacity of 145,000 tpa. These prices have been shown in Table A.21 below.

Table A.21: Key average metal product prices (present day 2007)

Component	Units	Value	Source
Cu	US$/tonne	$ 7,123.56	Datastream (2008) - daily average over 2007
Ag	US$/tonne	$ 472,033.24	Datastream (2008) - daily average over 2007 (converted from US$/oz)
Au	US$/tonne	$ 24,579,717.05	Datastream (2008) - daily average over 2007 (converted from US$/oz)

Primary cost estimates for each process were derived from 1994 data presented by Biswas and Davenport (1994). These were adjusted for inflation to the year 2007 using the Marshall & Swift cost index. These adjustment details are presented in Table A.22 below.

Table A.22: Assumptions made for inflation adjustments

Item		**Source**
Primary cost estimations:		Biswas and Davenport (1994): Extractive Metallurgy of Copper, 3rd Ed. (1994)
Working capital as % of initial capital investment	10%	Biswas and Davenport (1994)
Inflation index (1994)	993.4	Marshall & Swift index - http://www.eng-tips.com/viewthread.cfm?qid=78988&page=3
Inflation index (2007)	1362.7	Marshall & Swift index - http://goliath.ecnext.com/coms2/gi_0199-6601539/Marshall-Swift-equipment-cost-index.html

The adjusted capital and operating costs have been shown in Table A.23 below.

Table A.23: Inflation-adjusted capital and operating costs

Technology / Process	**Capital Cost**	Inflation adjusted	**Operating Cost**	Inflation adjusted
	US$/annual tonne Cu		**US$/tonne Cu**	
Mine (open-pit)	$ 1,000.00	$ 1,371.75	$ 200.00	$ 274.35
Concentrator	$ 2,000.00	$ 2,743.51	$ 300.00	$ 411.53
Electrorefining	$ 500.00	$ 685.88	$ 100.00	$ 137.18
Reverb. smelter (incl. acid plant)	$ 2,500.00	$ 3,429.38	$ 300.00	$ 411.53
Flash smelter (incl. acid plant)	$ 2,000.00	$ 2,743.51	$ 300.00	$ 411.53
HL-SX-EW	$ 1,440.00	$ 1,975.33	$ 370.00	$ 507.55
Sales & distribution			$ 50.00	$ 68.59
Local management / overheads			$ 100.00	$ 137.18
Finance costs			$ 900.00	$ 1,234.58

Key assumptions and data regarding the variability of the above costs with ore grade and throughput or plant capacity have been presented below.

1) Mining cost variability with ore grade

Source: Biswas and Davenport (1994)

Assumptions: Capital costs double from 0.5% to 1% ore grade, while operating costs are halved (Biswas & Davenport, 1994)
Linear relationship between mining cost and ore grade

Table A.24: Variation of mining capital and operating costs with ore grade

Ore grade	**Capital cost (US$/t)**	**Capital cost - inflation adjusted (US$/t)**	**Operating cost (US$/t)**	**Operating cost - inflation adjusted (US$/t)**
0.50%	$ 1,166.67	$ 1,600.38	$ 233.34	$ 320.08
0.60%	$ 1,100.00	$ 1,508.93	$ 220.00	$ 301.79
0.70%	$ 1,033.33	$ 1,417.48	$ 206.67	$ 283.50
0.80%	$ 966.66	$ 1,326.02	$ 193.34	$ 265.21
0.90%	$ 900.00	$ 1,234.57	$ 180.00	$ 246.92
1.00%	$ 833.33	$ 1,143.12	$ 166.67	$ 228.63

2) Concentrator cost variability with ore grade

Source: Biswas and Davenport (1994)

Assumptions: Capital costs double from 0.5% to 1% ore grade, while operating costs are halved (Biswas & Davenport, 1994)
Linear relationship between concentrator cost and ore grade

Table A.25: Variation of concentrator capital and operating costs with ore grade

Ore grade	**Capital cost (US$/t)**	**Capital cost - inflation adjusted (US$/t)**	**Operating cost (US$/t)**	**Operating cost - inflation adjusted (US$/t)**
0.50%	$ 2,333.34	$ 3,200.76	$ 350.00	$ 480.11
0.60%	$ 2,200.00	$ 3,017.86	$ 330.00	$ 452.68
0.70%	$ 2,066.67	$ 2,834.96	$ 310.00	$ 425.24

0.80%	$ 1,933.34	$ 2,652.06	$ 290.00	$ 397.81
0.90%	$ 1,800.00	$ 2,469.16	$ 270.00	$ 370.37
1.00%	$ 1,666.67	$ 2,286.26	$ 250.00	$ 342.94

3) Metal extraction cost variability with ore grade

Source: Biswas and Davenport (1994)

Table A. 26: Variation of leaching capital and operating costs with concentrate grade

Ore grade	Capital cost (US$/t)	Capital cost - inflation adjusted (US$/t)	Operating cost (US$/t)	Operating cost - inflation adjusted (US$/t)
0.50%	$ 2,906.52	$ 3,987.03	$ 746.45	$ 1,023.95
0.60%	$ 2,119.49	$ 2,907.42	$ 544.33	$ 746.68
0.70%	$ 1,622.85	$ 2,226.15	$ 416.78	$ 571.72
0.80%	$ 1,287.77	$ 1,766.50	$ 330.72	$ 453.67
0.90%	$ 1,050.12	$ 1,440.51	$ 269.69	$ 369.95
1.00%	$ 874.96	$ 1,200.23	$ 224.71	$ 308.24

NOTES:

- It has been assumed that smelting costs are governed by the concentrate grade, rather than the ore grade (Biswas and Davenport, 1994). As such, smelting costs have not been varied as a function of the ore grade above.
- Costs associated with finance, sales and distribution have also been assumed to not be strongly driven by the ore grade as overhead costs, and as such have not been varied in this analysis.

4) Capital cost variability with plant capacity (throughput)

It has been assumed that the 'six-tenths' rule can be used to approximate the variability of all types of capital costs within each of the three copper processing routes, i.e.

$$C_b = C_a \cdot \left(\frac{S_b}{S_a}\right)^{0.6}$$

Equation A.2: Formula describing the variation of capital costs with plant capacity

The results of the above performance assessment have been included in Table A.27 below.

Table A.27: Economic performance assessment results data

	Reverb. Smelting	Flash Smelting	HL-SX-EW
COSTS			
Total major equipment costs (US$/annual tonne Cu)	$ 8,230.52	$ 7,544.64	$ 3,347.08
Working capital (US$/annual tonne Cu)	$ 823.05	$ 754.46	$ 334.71
Total initial capital investment	$ 9,053.57	$ 8,299.11	$ 3,681.79
Total operating costs	$ 2,674.92	$ 2,674.92	$ 2,222.24
REVENUE			
Cu sales (US$/tonne Cu)	$ 7,123.56	$ 7,123.56	$ 7,123.56
Ag credits (US$/tonne Cu)	$ 2,291.62	$ 2,157.51	$ -

Au credits (US$/tonne Cu)	$ 2,983.23	$ 2,808.65	$ -
Total revenue (US$/tonne Cu)	$ 12,398.41	$ 12,089.72	$ 7,123.56
VALUE ADD	**$ 669.92**	**$ 1,115.69**	**$ 1,219.54**

The eco-efficiency indicators were then computed as the ratio of the economic value add in Table A.27 above to the environmental damage as shown in Appendix A.1. They were also normalised as described in Chapter 4 to compute relative (graphical) eco-efficiency performance scores. These indicators and performance scores are shown in Table A.28 and Table A.29 below.

Table A.28: Numeric eco-efficiency indicators for the copper processing alternatives

Alternatives	***Eco-efficiency indicators***			
	GHG emissions	**Water consumption**	**Ecotoxicity**	**Resource depletion**
	US$/kg CO_2	US$/$m^3$ H_2O	US$/ton 1,4-DB eq.	US$/ton Sb eq.
Reverb	75.0	6.0	0.0015	12566
Flash	96.3	11.2	0.0026	25239
HL-SX-EW	102.5	7.3	0.0007	18599

Table A.29: Relative graphical eco-efficiency performance scores for the copper processing alternatives

Alternatives	***Criteria***				
	GHG emissions	**Water consumption**	**Ecotoxicity**	**Resource depletion**	**Economic**
Reverb	0.91	0.88	0.50	0.98	0.67
Flash	0.71	0.79	0.47	0.81	1.11
HL-SX-EW	1.28	1.32	2.03	1.21	1.22

B.2 Case Study 2

Capital and operating costs were estimated from www.matche.com at 2006 prices (Mwakyusa, 2007). A cost summary for all dewatering equipment types used is provided in Table A.30 below.

Table A.30: Capital and operating cost summary for dewatering circuit technologies (Mwakyusa, 2007)

Item / Unit	Cost (US$)	
Cyclone unit capital cost	$	42,200
Conventional thickener unit capital cost	$	230,959
High-rate thickner unit capital cost	$	115,479
Filter unit capital cost	$	492,253
Tailings dam capital cost	$	4,550,000
Hydrocyclone operating cost (US$/t)	$	0.12
Conventional thickener operating cost (US$/t)	$	0.42
High-rate thickener operating cost (US$/t)	$	0.21
Filter operating cost (US$/t)	$	0.30
Wet tailings dam capital US$/ m^2	$	6.50
Wet tailings dam operating cost US$/t	$	0.60

Based on the above unit costs, the overall capital and operating costs for each dewatering alternative were then computed as shown in Table A.31 below.

Table A.31: Capital and operating cost summary for the dewatering circuit alternatives

	Capital (US$)				Operating/year (US$)				Tailings Dam Cost (US$)		Total Cost (US$)	
Options	Cyclones	Conventional thickener	High-rate thickener	Filter	Cyclones	Conventional thickener	High-rate thickener	Filter	Capital	Operating/yr	Capital	Operating/year
1		461,918				1,398,096			4,550,000	1,997,280	5,011,918	3,395,376
2			230,959				699,048		4,044,444	1,997,280	4,275,403	2,696,328
3		461,918		6,891,542		1,398,096		998,640	3,589,444	665,760	10,942,904	3,062,496
4			230,959	6,891,542			699,048	998,640	3,589,444	466,032	10,711,945	2,163,720
5	337,600	230,959		4,922,530	399,456	501,750		658,080	4,030,588	665,760	9,521,677	2,225,046
6	337,600		115,480	4,922,530	399,456		426,741	658,080	3,696,146	665,760	9,071,755	2,150,037

Cost savings enjoyed by each design alternatives (i.e. the economic benefit) were due to the additional water and cyanide recovered were calculated assuming 2006 water and cyanide prices. These, and other key supporting assumptions, have been shown in Table A.32 below.

Table A.32: Assumptions for cyanide and water recovery as cost savings (from Mwakyusa, 2007)

Average cyanide concentration in gold CIL circuit		280	ppm
Average cyanide concentration in the tailings stream		260	ppm
Average cyanide loss through UV rays		30	%
Cyanide concentration in the tailings stream		182	ppm
Cyanide price per tonne (Global)	$	2,200.00	/tonne
Water price US$/tonne (Tanzania)	$	0.35	/tonne

These derived savings have been reported on a mass and US$ basis in Table A.33 below.

Table A.33: Water and cyanide cost savings from dewatering circuit alternatives

Option	Final % Solids	Water Recovered (t/h)	Volume (m^3/h)	Total CN Ton/year	Cyanide cost savings / annum	Water cost savings / annum
1	60.0	133.3	133.33	403.89	$ 888,568	$ 388,360
2	70.0	228.6	228.57	692.39	$ 1,523,259	$ 665,760
3	79.0	293.7	293.67	889.59	$ 1,957,098	$ 855,375
4	79.0	293.7	293.67	889.59	$ 1,957,098	$ 855,375
5	70.3	230.8	230.80	699.14	$ 1,538,112	$ 672,252
6	76.9	279.8	279.77	847.49	$ 1,864,476	$ 814,893

The eco-efficiency indicators were then computed as the ratio of the cost savings in Table A.33 above to the water loss as shown in Appendix A.2. These indicators are shown in Table A.34 below.

Table A.34: Summary of numeric eco-efficiency indicators and (absolute and relative) graphical eco-efficiency scores

Options	NPV	Dissipative Water Loss over discount period (tonnes)	Water eco-efficiency (US$/ton H2O loss)	Relative dissipative water Loss indicator	Relative economic value add indicator	Relative economic value add indicator (corrected)
1	$ -15,424,636	11,489,280	-1.34	1.19	1.38	0.62
2	$ -8,079,430	10,004,846	-0.81	1.04	0.72	1.28
3	$ -14,024,057	8,734,163	-1.61	0.90	1.26	0.74
4	$ -9,752,368	8,734,163	-1.12	0.90	0.87	1.13
5	$ -11,125,797	9,965,368	-1.12	1.03	1.00	1.00
6	$ -8,556,455	9,027,815	-0.95	0.93	0.77	1.23

APPENDIX C

Distinguishability Analyses

Further methodological details on the computation of distinguishability indices for the considered design alternatives are provided in this section.

C.1 Methodological details

If ***A*** is the set of *n* design alternatives under consideration, ***X*** is the set of attributes used to measure the performance of the alternatives (i.e. performance values) and ***G*** is the set of performance criteria, then sets ***A***, ***X*** and ***G*** could be represented as follows:

$$\boldsymbol{A} = \{a_1 ...., a_i, ..., a_n\} \tag{3.1}$$

$$\boldsymbol{G} = \{g_1 ...., g_j, ..., g_n\} \tag{3.2}$$

$$\boldsymbol{X} = \{x_1 ...., x_j, ..., x_n\} \tag{3.3}$$

Equation A.3: Mathematical representations of the design alternatives, performance criteria and performance values

The performance of an alternative a_i in criterion g_j can thus be represented as $g_j(a_i)$ or x_{ij}. The data required for the calculation of the distinguishability index are a *best guess* (x_{ij}), a *likely maximum* (x^+_{ij}) and a *likely minimum* (x^-_{ij}) value for the performance scores of each alternative a_i for each performance criterion *j*. The difference between the best guess and the likely maximum and that between the best guess and the likely minimum form the *positive dispersion threshold* (U^+_{ij}) and *negative dispersion threshold* (U^-_{ij}) respectively. These are the likely positive and negative off-sets from the best guess performance value due to uncertainty, and can be expressed as follows:

$$\mathbf{U^+_{ij} = x^+_{ij} - x_{ij}} \tag{4.1}$$

$$\mathbf{U^-_{ij} = x_{ij} - x^-_{ij}} \tag{4.2}$$

Equation A.4: Mathematical representations of the positive and negative dispersion thresholds for a performance value

Figure A.2 below shows these key criteria diagrammatically.

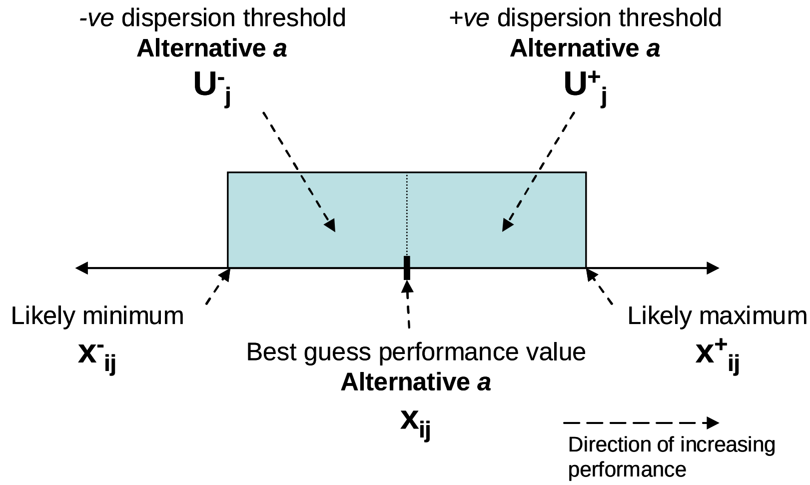

Figure A.2: Key distinguishability criteria for the performance of a design alternative *a*

The required minimum difference between the best estimates of the performance values of two alternatives (say *a* and *b*) for a particular performance criterion *j* which ensures that the alternatives are distinguishable from one another is referred to as the *distinguishability threshold* $v_j(a,b)$. If $x_j(a) > x_j(b)$, then the distinguishability threshold can be defined as

$$v(a,b) = U_j^-(a) + U_j^+(b)$$

Equation A.5: Mathematical definition of a distinguishability threshold

The distinguishability threshold concept is illustrated in Figure A.3 below.

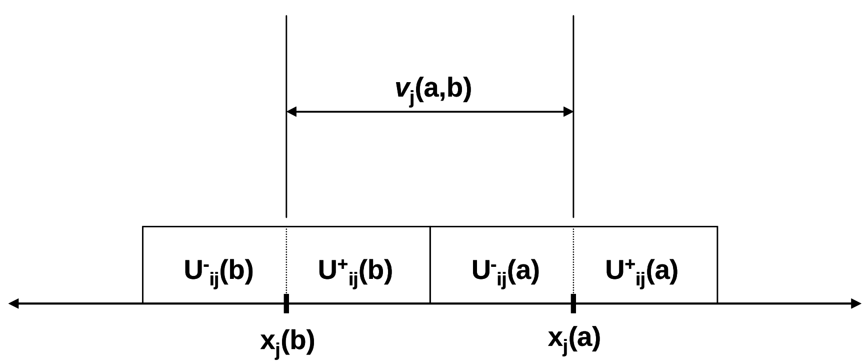

Figure A.3: Distinguishability threshold for criterion *j* for two alternatives *a* and *b*

The alternatives are therefore distinguishable if the difference between their best estimates $x_j(a)$ and $x_j(b)$ *exceeds* the distinguishability threshold $v_j(a,b)$, i.e. if

$$| x_j(a) - x_j(b) | > v_j(a,b),$$

Equation A.6: The minimum criterion for distinguishability between performance values of two alternatives

This has been shown diagrammatically in Figure A.4 below.

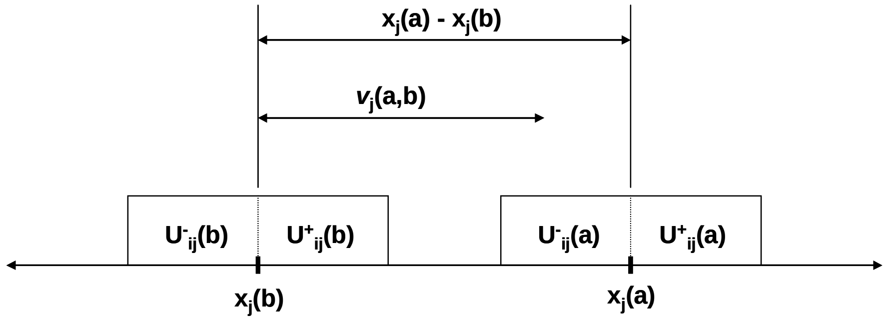

Figure A.4: A graphical depiction of a distinguishability assessment

The alternatives can be compared in a pairwise manner for each criterion to determine whether they are distinguishable from each other. An alternative is therefore defined as completely distinguishable from another when the best estimate difference far exceeds the distinguishability threshold, and completely indistinguishable when the best estimate difference is much less than the distinguishability threshold. In quantifying this assessment, a value can be assigned to a distinguishability parameter $d_j(a,b)$ to specify whether two alternatives are distinguishable from each other when considering a particular performance criterion. This distinguishability parameter can be assigned a number 1 to indicate complete distinguishability, and a value of 0 for indistinguishability. Since an indication is required of whether the alternatives are distinguishable from each other considering all the performance criteria, the information regarding distinguishability can be aggregated across the criteria into a distinguishability index (DI) D(a,b) for each pairwise comparison of alternatives, as illustrated by Equation A.6 below.

$$D(a,b) = \frac{\sum_{j=1}^{n} d_j(a,b)}{n}$$

Equation A.7: Mathematical formula for distinguishability indices for pairwise comparisons of alternatives

In a similar manner, the values for the DI can then be aggregated across all the pairwise comparisons into a single score for each of the alternatives considered, the aggregated distinguishability index (ADI). A value of 0 for the ADI of a certain alternative implies complete indistinguishability from all other alternatives, while a value of 1 indicates that an alternative is completely distinguishable from other alternatives. Intermediate values imply ‘weak’ or ‘partial’ distinguishability. Figure A.5 below shows a summary of this methodology as developed by Basson (2004).

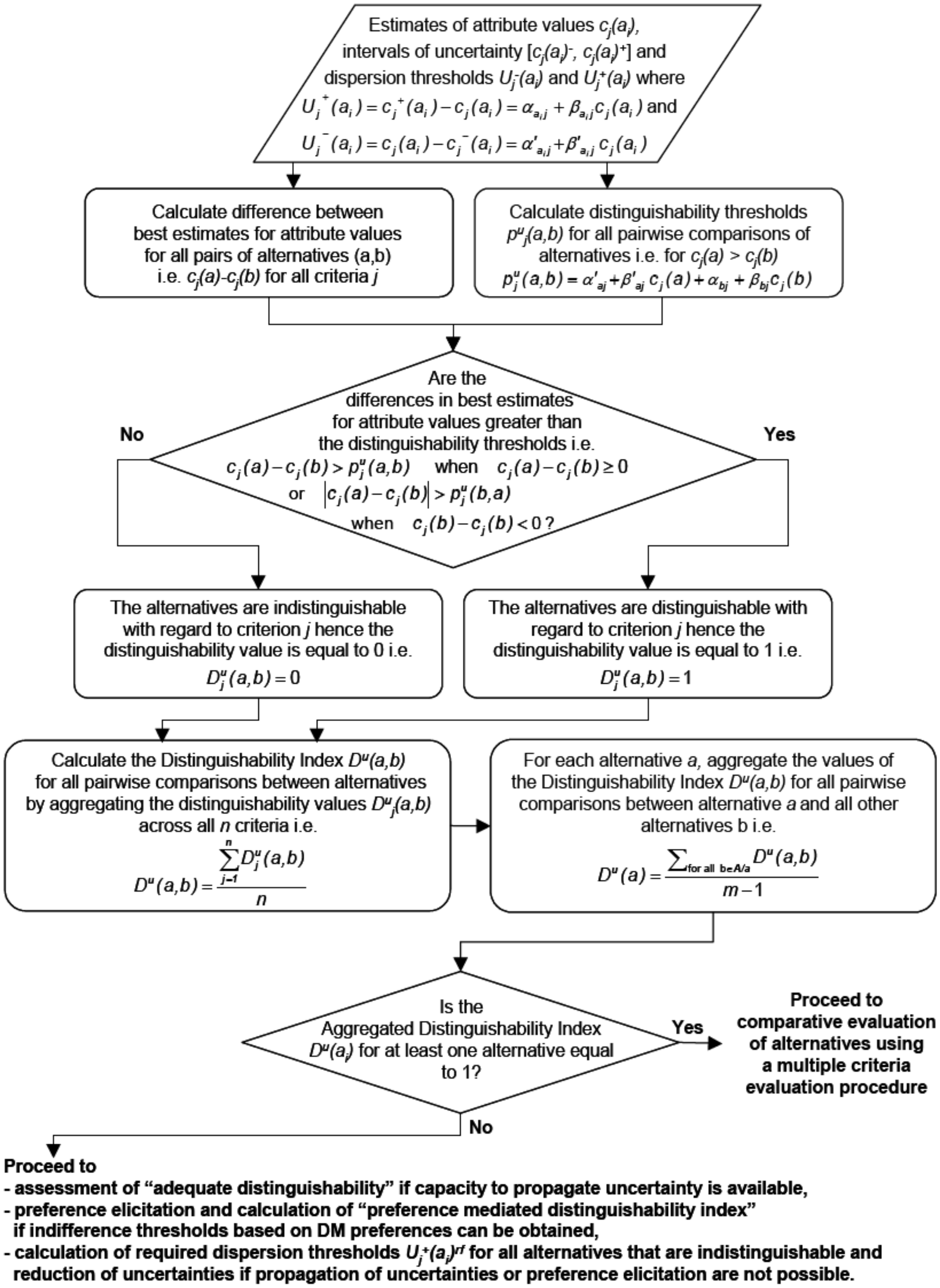

Figure A.5: Overview flowchart of the distinguishability approach by Basson (2004)

Distinguishability analysis data have been included for each case study investigated in this thesis in section C.2 and section C.3 below.

C.2 Distinguishability Data – Case Study 1

Table A.35: Uncertainties in the attribute values and dispersion thresholds for numeric eco-efficiency indicators (Case study 1)

Alternatives	Likely Max.	Likely Min.	-ve dispersion threshold	+ve dispersion threshold
GHG emissions				
Reverb	123.7	26.2	48.7	48.7
Flash	158.9	33.7	62.6	62.6
HL-SX-EW	146.7	31.1	57.8	57.8
Water consumption				
Reverb	9.98	2.12	3.93	3.93
Flash	18.49	3.92	7.29	7.29
HL-SX-EW	12.10	2.57	4.77	4.77
Ecotoxicity				
Reverb	0.0025	0.0005	0.0010	0.0010
Flash	0.0043	0.0009	0.0017	0.0017
HL-SX-EW	0.0011	0.0002	0.0004	0.0004
Resource efficiency				
Reverb	20734	4398	8168	8168
Flash	41644	8834	16405	16405
HL-SX-EW	30688	6510	12089	12089

Table A.36: Eco-efficiency indicator distinguishability thresholds and best estimate differences (Case study 1)

Distinguishability thresholds				**Best estimate differences**			
GHG emissions				***GHG emissions***			
	Reverb	Flash	HL-SX-EW		Reverb	Flash	HL-SX-EW
Reverb		111	107	Reverb		-21	-14
Flash	111		120	Flash	21		7
HL-SX-EW	107	120		HL-SX-EW	14	-7	
Water consumption				***Water consumption***			
Reverb		11	9	Reverb		-5	-1
Flash	11		12	Flash	5		4
HL-SX-EW	9	12		HL-SX-EW	1	-4	
Ecotoxicity				***Ecotoxicity***			
Reverb		0.003	0.001	Reverb		-0.001	0.001
Flash	0.003		0.002	Flash	0.001		0.002
HL-SX-EW	0.001	0.002		HL-SX-EW	-0.001	-0.002	
Resource efficiency				***Resource efficiency***			
Reverb		24573	20257	Reverb		-12673	-6033
Flash	24573		28494	Flash	12673		6640
HL-SX-EW	20257	28494		HL-SX-EW	6033	-6640	

Table A.37: Numeric eco-efficiency distinguishability indicators for the copper process alternatives (Case study 1)

Distinguishability indicator			
GHG emissions			
	Reverb	Flash	HL-SX-EW
Reverb		0	0
Flash	0		0
HL-SX-EW	0	0	
Water consumption			
Reverb		0	0
Flash	0		0
HL-SX-EW	0	0	
Ecotoxicity			
Reverb		0	0
Flash	0		0
HL-SX-EW	0	0	
Resource efficiency			
Reverb		0	0
Flash	0		0
HL-SX-EW	0	0	

Table A.38: Numeric eco-efficiency aggregated distinguishability indices for the copper process alternatives (Case study 1)

	Reverb	Flash	HL-SX-EW	**ADI**
Reverb		0	0	**0.0**
Flash	0		0	**0.0**
HL-SX-EW	0	0		**0.0**

Table A.39: Uncertainties in the attribute values and dispersion thresholds for relative graphical eco-efficiency (Case study 1)

Alternatives	**Likely Max.**	**Likely Min.**	**-ve dispersion threshold**	**+ve dispersion threshold**
GHG emissions				
Reverb	1.1	0.7	0.2	0.2
Flash	0.9	0.5	0.2	0.2
HL-SX-EW	1.7	1.0	0.3	0.3
Water consumption				
Reverb	1.10	0.66	0.22	0.22
Flash	0.99	0.59	0.20	0.20
HL-SX-EW	1.66	0.99	0.33	0.33
Eco-toxicity				
Reverb	0.63	0.38	0.13	0.13
Flash	0.59	0.36	0.12	0.12
HL-SX-EW	2.53	1.52	0.51	0.51
Resource efficiency				
Reverb	1.23	0.74	0.25	0.25
Flash	1.02	0.61	0.20	0.20
HL-SX-EW	1.51	0.90	0.30	0.30
Economic performance				
Reverb	0.94	0.40	0.27	0.27
Flash	1.56	0.67	0.45	0.45
HL-SX-EW	1.70	0.73	0.49	0.49

Table A.40: Relative graphical eco-efficiency distinguishability thresholds and best estimate differences (Case study 1)

Distinguishability thresholds				**Best estimate differences**			
GHG emissions				***GHG emissions***			
	Reverb	Flash	HL-SX-EW		Reverb	Flash	HL-SX-EW
Reverb		0.40	0.57	Reverb		0.2	-0.5
Flash	0.40		0.52	Flash	-0.2		-0.7
HL-SX-EW	0.57	0.52		HL-SX-EW	0.5	0.7	
Water consumption				***Water consumption***			
Reverb		0.42	0.55	Reverb		0.1	-0.4
Flash	0.42		0.53	Flash	-0.1		-0.5
HL-SX-EW	0.55	0.53		HL-SX-EW	0.4	0.5	
Ecotoxicity				***Ecotoxicity***			
Reverb		0.24	0.63	Reverb		0.03	-1.53
Flash	0.24		0.62	Flash	-0.03		-1.55
HL-SX-EW	0.63	0.62		HL-SX-EW	1.53	1.55	
Resource efficiency				***Resource efficiency***			
Reverb		0.45	0.55	Reverb		0.2	-0.2
Flash	0.45		0.50	Flash	-0.2		-0.4
HL-SX-EW	0.55	0.50		HL-SX-EW	0.2	0.4	
Economic performance				***Economic performance***			
Reverb		0.71	0.75	Reverb		-0.4	-0.5
Flash	0.71		0.93	Flash	0.4		-0.1
HL-SX-EW	0.75	0.93		HL-SX-EW	0.5	0.1	

Table A.41: Relative graphical eco-efficiency distinguishability indicators (Case study 1)

Distinguishability indicator			
GHG emissions			
	Reverb	Flash	HL-SX-EW
Reverb		0	0
Flash	0		1
HL-SX-EW	0	1	
Water consumption			
Reverb		0	0
Flash	0		1
HL-SX-EW	0	1	
Ecotoxicity			
Reverb		0	1
Flash	0		1
HL-SX-EW	1	1	
Resource efficiency			
Reverb		0	0
Flash	0		0
HL-SX-EW	0	0	
Economic performance			
Reverb		0	0
Flash	0		0
HL-SX-EW	0	0	

Table A.42: Relative graphical eco-efficiency aggregated distinguishability indices for the copper process alternatives

	Reverb	Flash	HL-SX-EW	**ADI**
Reverb		0	0.25	**0.13**
Flash	0		0.75	**0.38**
HL-SX-EW	0.25	0.75		**0.50**

C.3 Distinguishability Data – Case Study 2

Table A.43: Uncertainties in the attribute values and dispersion thresholds for numeric eco-efficiency indicators (Case study 2)

Option	Likely Max.	Likely Min.	-ve dispersion threshold	+ve dispersion threshold
1	-1.101	-1.584	0.242	0.242
2	-0.662	-0.953	0.145	0.145
3	-1.317	-1.895	0.289	0.289
4	-0.916	-1.318	0.201	0.201
5	-0.915	-1.317	0.201	0.201
6	-0.777	-1.118	0.171	0.171

Table A.44: Eco-efficiency indicator distinguishability thresholds and best estimate differences (Case study 2)

	Distinguishability thresholds						Best estimate differences					
OPTION	**1**	**2**	**3**	**4**	**5**	**6**	**1**	**2**	**3**	**4**	**5**	**6**
1		0.39	0.53	0.44	0.44	0.41		-0.53	0.26	-0.23	-0.23	-0.39
2	0.39		0.43	0.35	0.35	0.32	0.53		0.80	0.31	0.31	0.14
3	0.53	0.43		0.49	0.49	0.46	-0.26	-0.80		-0.49	-0.49	-0.66
4	0.44	0.35	0.49		0.40	0.37	0.23	-0.31	0.49		0.00	-0.17
5	0.44	0.35	0.49	0.40		0.37	0.23	-0.31	0.49	0.00		-0.17
6	0.41	0.32	0.46	0.37	0.37		0.39	-0.14	0.66	0.17	0.17	

Table A.45: Numeric eco-efficiency distinguishability indicators and aggregated distinguishability index (Case study 2)

Option	1	2	3	4	5	6	ADI
1		1	0	0	0	0	**0.20**
2	1		1	0	0	0	**0.40**
3	0	1		0	0	1	**0.40**
4	0	0	0		0	0	**0.00**
5	0	0	0	0		0	**0.00**
6	0	0	1	0	0		**0.20**

Table A.46: Uncertainties in the attribute values and dispersion thresholds for absolute graphical eco-efficiency (Case study 2)

Alternatives	Likely Max.	Likely Min.	-ve dispersion threshold	+ve dispersion threshold
Net Present Value				
1	$ -17,275,592.00	$-13,573,679.43	$ 1,850,956.29	$ 1,850,956.29
2	$ -9,048,961.45	$ -7,109,898.28	$ 969,531.58	$ 969,531.58
3	$ -15,706,943.87	$-12,341,170.18	$ 1,682,886.84	$ 1,682,886.84
4	$ -10,922,652.30	$ -8,582,083.95	$ 1,170,284.17	$ 1,170,284.17
5	$ -12,460,893.20	$ -9,790,701.80	$ 1,335,095.70	$ 1,335,095.70
6	$ -9,583,229.73	$ -7,529,680.50	$ 1,026,774.61	$ 1,026,774.61
Water loss				
1	1.29E+07	1.01E+07	1.38E+06	1.38E+06
2	1.12E+07	8.80E+06	1.20E+06	1.20E+06
3	9.78E+06	7.69E+06	1.05E+06	1.05E+06
4	9.78E+06	7.69E+06	1.05E+06	1.05E+06
5	1.12E+07	8.77E+06	1.20E+06	1.20E+06
6	1.01E+07	7.94E+06	1.08E+06	1.08E+06

Table A.47: Absolute graphical eco-efficiency distinguishability thresholds (Case study 2)

Option	1	2	3	4	5	6
Net Present Value						
1		$ 2,820,487.87	$ 3,533,843.13	$ 3,021,240.46	$ 3,186,051.99	$ 2,877,730.90
2	$ 2,820,487.87		$ 2,652,418.43	$ 2,139,815.76	$ 2,304,627.28	$ 1,996,306.20
3	$ 3,533,843.13	$ 2,652,418.43		$ 2,853,171.02	$ 3,017,982.54	$ 2,709,661.46
4	$ 3,021,240.46	$ 2,139,815.76	$ 2,853,171.02		$ 2,505,379.87	$ 2,197,058.79
5	$ 3,186,051.99	$ 2,304,627.28	$ 3,017,982.54	$ 2,505,379.87		$ 2,361,870.31
6	$ 2,877,730.90	$ 1,996,306.20	$ 2,709,661.46	$ 2,197,058.79	$ 2,361,870.31	
Water loss						
1		2.58E+06	2.43E+06	2.43E+06	2.57E+06	2.46E+06
2	2.58E+06		2.25E+06	2.25E+06	2.40E+06	2.28E+06
3	2.43E+06	2.25E+06		2.10E+06	2.24E+06	2.13E+06
4	2.43E+06	2.25E+06	2.10E+06		2.24E+06	2.13E+06
5	2.57E+06	2.40E+06	2.24E+06	2.24E+06		2.28E+06
6	2.46E+06	2.28E+06	2.13E+06	2.13E+06	2.28E+06	

Table A.48: Absolute graphical eco-efficiency best estimate differences (Case study 2)

Option	1	2	3	4	5	6
	Net Present Value					
1		$ - 7,345,205.85	$ - 1,400,578.69	$ - 5,672,267.59	$ - 4,298,838.22	$ - 6,868,180.60
2	$ 7,345,205.85		$ 5,944,627.16	$ 1,672,938.25	$ 3,046,367.63	$ 477,025.25
3	$ 1,400,578.69	$ - 5,944,627.16		$ - 4,271,688.90	$ - 2,898,259.53	$ - 5,467,601.91
4	$ 5,672,267.59	$ - 1,672,938.25	$ 4,271,688.90		$ 1,373,429.37	$ - 1,195,913.01
5	$ 4,298,838.22	$ - 3,046,367.63	$ 2,898,259.53	$ - 1,373,429.37		$ - 2,569,342.38
6	$ 6,868,180.60	$ - 477,025.25	$ 5,467,601.91	$ 1,195,913.01	$ 2,569,342.38	
Option	***Water loss***					
1		1.48E+06	2.76E+06	2.76E+06	1.52E+06	2.46E+06
2	-1.48E+06		1.27E+06	1.27E+06	3.95E+04	9.77E+05
3	-2.76E+06	-1.27E+06		0.00E+00	-1.23E+06	-2.94E+05
4	-2.76E+06	-1.27E+06	0.00E+00		-1.23E+06	-2.94E+05
5	-1.52E+06	-3.95E+04	1.23E+06	1.23E+06		9.38E+05
6	-2.46E+06	-9.77E+05	2.94E+05	2.94E+05	-9.38E+05	

Table A.49: Absolute graphical eco-efficiency distinguishability indicators (Case study 2)

Option	1	2	3	4	5	6
Net Present Value						
1		1	0	1	1	1
2	1		1	0	1	0
3	0	1		1	0	1
4	1	0	1		0	0
5	1	1	0	0		1
6	1	0	1	0	1	
Water loss						
1		0	1	1	0	0
2	0		0	0	0	0
3	1	0		0	0	0
4	1	0	0		0	0
5	0	0	0	0		0
6	0	0	0	0	0	

Table A.50: Absolute graphical eco-efficiency aggregated distinguishability indices (Case study 2)

Option	1	2	3	4	5	6	ADI
1		0.50	0.50	1.00	0.50	0.50	**0.60**
2	0.50		0.50	0.00	0.50	0.00	**0.30**
3	0.50	0.50		0.50	0.00	0.50	**0.40**
4	1.00	0.00	0.50		0.00	0.00	**0.30**
5	0.50	0.50	0.00	0.00		0.50	**0.30**
6	0.50	0.00	0.50	0.00	0.50		**0.30**

APPENDIX D

Sensitivity Analyses

The raw data for the quantitative sensitivity analyses performed for each case study in this thesis have been provided in this section.

D.1 Case Study 1

Table A.51: Variation of environmental impacts with ore grade for the copper process alternatives

Technology	GHG Emissions	Water Consumption	Eco-toxicity		Resource Depletion	
0.50%						
	kg CO2-equiv.	tonnes	Min	Max	Min	Max
Reverb. [Guirco (2005)]	483808.3	21115136.7				
Normalised /tonne Cu	3.3	145.3	10169	593501	0.002	0.047
Flash [Guirco (2005)]	312056.7	20156176.1				
Normalised /tonne Cu	2.2	139.3	10258	598712	0.002	0.047
HLSXEW [Guirco (2005)]	360888.9	31630685.4				
Normalised /tonne Cu	2.5	218.1	107438	2389911	0.002	0.070
0.60%						
Technology	**GHG Emissions**	**Water Consumption**	**Eco-toxicity**		**Resource Depletion**	
	kg CO2-equiv.	tonnes	Min	Max	Min	Max
Reverb. [Guirco (2005)]	447699.5	17599047.8				
Normalised /tonne Cu	3.1	121.1	8448	493078	0.001	0.039

Flash [Guirco (2005)]	273941.9	16799909.4				
Normalised /tonne Cu	1.9	116.1	8530	497839	0.001	0.039
HLSXEW [Guirco (2005)]	300740.7	26359456.0				
Normalised /tonne Cu	2.1	181.8	89569	1992430	0.002	0.058

0.70%

Technology	GHG Emissions kg CO2-equiv.	Water Consumption tonnes	Eco-toxicity Min	Max	Resource Depletion Min	Max
Reverb. [Guirco (2005)]	432262.1	16095827.3				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	246717.0	14402576.1				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	275025.8	24105870.8				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053

0.80%

Technology	GHG Emissions kg CO2-equiv.	Water Consumption tonnes	Eco-toxicity Min	Max	Resource Depletion Min	Max
Reverb. [Guirco (2005)]	402563.5	13203936.7				
Normalised /tonne Cu	2.8	90.8	6297	367549	0.001	0.029
Flash [Guirco (2005)]	226298.3	12604576.1				
Normalised /tonne Cu	1.6	87.1	6369	371749	0.001	0.029
HLSXEW [Guirco (2005)]	225555.5	19770428.3				
Normalised /tonne Cu	1.6	136.3	67233	1495578	0.001	0.044

0.90%

Technology	GHG Emissions kg CO2-equiv.	Water Consumption tonnes	Eco-toxicity Min	Max	Resource Depletion Min	Max
Reverb. [Guirco (2005)]	387518.2	11738899.6				
Normalised /tonne Cu	2.7	80.8	5580	325706	0.001	0.026

Technology	GHG Emissions kg CO2-equiv.	Water Consumption tonnes	Eco-toxicity Min	Max	Resource Depletion Min	Max
Flash [Guirco (2005)]	210417.2	11206131.7				
Normalised /tonne Cu	1.5	77.4	5649	329718	0.001	0.026
HLSXEW [Guirco (2005)]	200493.8	17574085.2				
Normalised /tonne Cu	1.4	121.2	59788	1329961	0.001	0.039
			1.0%			
Technology	**GHG Emissions** **kg CO2-equiv.**	**Water Consumption** **tonnes**	**Eco-toxicity** **Min**	**Max**	**Resource Depletion** **Min**	**Max**
Reverb. [Guirco (2005)]	375481.9	10566870.0				
Normalised /tonne Cu	2.6	72.7	5007	292232	0.001	0.023
Flash [Guirco (2005)]	197712.2	10087376.1				
Normalised /tonne Cu	1.4	69.7	5073	296094	0.001	0.023
HLSXEW [Guirco (2005)]	180444.4	15817010.1				
Normalised /tonne Cu	1.2	109.1	53832	1197467	0.001	0.035

Table A.52: Variation of reverbaratory smelting costs with ore grade

Ore grade	Mine & Concentrator	Smelting & Refining		Sales, distribution, overheads, finance	**TOTAL**
	Capital/Op.	Capital	Operating	Operating	
0.50%	$ 5,601.33	$ 4,115.26	$ 548.70	$ 1,440.34	**$ 12,597.27**
0.60%	$ 5,281.25	$ 4,115.26	$ 548.70	$ 1,440.34	**$ 12,249.76**
0.70%	$ 4,961.18	$ 4,115.26	$ 548.70	$ 1,440.34	**$ 11,902.25**
0.80%	$ 4,641.10	$ 4,115.26	$ 548.70	$ 1,440.34	**$ 11,554.74**
0.90%	$ 4,321.03	$ 4,115.26	$ 548.70	$ 1,440.34	**$ 11,207.23**
1.00%	$ 4,000.95	$	$ 548.70	$ 1,440.34	**$ 10,859.72**

		4,115.26			

Table A.53: Variation of flash smelting costs with ore grade

Ore grade	Mine & Concentrator	Smelting & Refining		Sales, distribution, overheads, finance	**TOTAL**
	Capital/Op.	Capital	Operating	Operating	
0.50%	$ 5,601.33	$ 3,429.38	$ 548.70	$ 1,440.34	**$11,911.40**
0.60%	$ 5,281.25	$ 3,429.38	$ 548.70	$ 1,440.34	**$11,563.89**
0.70%	$ 4,961.18	$ 3,429.38	$ 548.70	$ 1,440.34	**$11,216.37**
0.80%	$ 4,641.10	$ 3,429.38	$ 548.70	$ 1,440.34	**$10,868.86**
0.90%	$ 4,321.03	$ 3,429.38	$ 548.70	$ 1,440.34	**$10,521.35**
1.00%	$ 4,000.95	$ 3,429.38	$ 548.70	$ 1,440.34	**$10,173.84**

Table A.54: Variation of HL-SX-EW costs with ore grade

Ore grade	Mine	HL-SX-EW		Sales, distribution, overheads, finance	**TOTAL**
	Capital/Op.	Capital	Operating	Operating	
0.50%	$ 1,920.46	$ 3,987.03	$ 1,023.95	$ 1,440.34	**$ 8,371.77**
0.60%	$ 1,810.71	$ 2,907.42	$ 746.68	$ 1,440.34	**$ 6,905.15**
0.70%	$ 1,700.97	$ 2,226.15	$ 571.72	$ 1,440.34	**$ 5,939.19**
0.80%	$ 1,591.23	$ 1,766.50	$ 453.67	$ 1,440.34	**$ 5,251.75**
0.90%	$ 1,481.49	$ 1,440.51	$ 369.95	$ 1,440.34	**$ 4,732.30**
1.00%	$ 1,371.75	$ 1,200.23	$ 308.24	$ 1,440.34	**$ 4,320.57**

Table A.55: Variation of environmental impacts with plant throughput for the copper process alternatives

100,000 tpa						
Technology	**GHG Emissions kg CO2-equiv.**	**Water Consumption tonnes**	**Eco-toxicity Min**	**Max**	**Resource Depletion Min**	**Max**
Reverb. [Guirco (2005)]	297418.9	11074769.9				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	170513.6	9954058.3				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	189673.0	16624740.0				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053
120,000 tpa						
Technology	**GHG Emissions kg CO2-equiv.**	**Water Consumption tonnes**	**Eco-toxicity Min**	**Max**	**Resource Depletion Min**	**Max**
Reverb. [Guirco (2005)]	356902.6	13289723.8				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	204616.4	11944870.0				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	227607.6	19949688.0				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053
140,000 tpa						
Technology	**GHG Emissions kg CO2-equiv.**	**Water Consumption tonnes**	**Eco-toxicity Min**	**Max**	**Resource Depletion Min**	**Max**
Reverb. [Guirco (2005)]	416386.4	15504677.8				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	238719.1	13935681.6				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	265542.2	23274636.0				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053
160,000 tpa						
Technology	**GHG Emissions**	**Water Consumption**	**Eco-toxicity**		**Resource Depletion**	

	kg CO2-equiv.	tonnes	Min	Max	Min	Max
Reverb. [Guirco (2005)]	475870.2	17719631.8				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	272821.8	15926493.3				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	303476.8	26599584.1				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053
180,000 tpa						
Technology	**GHG Emissions kg CO2-equiv.**	**Water Consumption tonnes**	**Eco-toxicity Min**	**Max**	**Resource Depletion Min**	**Max**
Reverb. [Guirco (2005)]	535354.0	19934585.7				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	306924.5	17917305.0				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	341411.4	29924532.1				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053
200,000 tpa						
Technology	**GHG Emissions kg CO2-equiv.**	**Water Consumption tonnes**	**Eco-toxicity Min**	**Max**	**Resource Depletion Min**	**Max**
Reverb. [Guirco (2005)]	594837.7	22149539.7				
Normalised /tonne Cu	3.0	110.7	7713	450145	0.001	0.035
Flash [Guirco (2005)]	341027.3	19908116.6				
Normalised /tonne Cu	1.7	99.5	7295	425788	0.001	0.033
HLSXEW [Guirco (2005)]	379346.0	33249480.1				
Normalised /tonne Cu	1.9	166.2	81930	1822496	0.002	0.053

Table A.56: Copper process alternatives annual revenue estimates as a function of plant capacity

Capacity (tpa)	REVERB.	FLASH	HL-SX-EW
100,000	$ 1,239,841,290	$ 1,208,972,299	$ 712,356,418
120,000	$ 1,487,809,548	$ 1,450,766,758	$ 854,827,701
140,000	$ 1,735,777,806	$ 1,692,561,218	$ 997,298,985
160,000	$ 1,983,746,064	$ 1,934,355,678	$ 1,139,770,268
180,000	$ 2,231,714,322	$ 2,176,150,137	$ 1,282,241,552
200,000	$ 2,479,682,580	$ 2,417,944,597	$ 1,424,712,835

Table A.57: Variation of capital and operating costs with throughput

	REVERB. SMELTING			FLASH SMELTING			HL-SX-EW		
Capacity (tpa)	Capital (UNIT)	Operating (UNIT)	OVERALL	Capital (UNIT)	Operating (UNIT)	OVERALL	Capital (UNIT)	Operating (UNIT)	OVERALL
100,000	$ 7,244	$ 2,675	$ 991,926,630	$ 6,641	$ 2,675	$ 931,557,073	$ 2,946	$ 2,222	$ 516,827,517
120,000	$ 8,082	$ 2,675	$ 1,290,806,076	$ 7,408	$ 2,675	$ 1,209,988,097	$ 3,287	$ 2,222	$ 661,060,629
140,000	$ 8,865	$ 2,675	$ 1,615,581,055	$ 8,126	$ 2,675	$ 1,512,156,694	$ 3,605	$ 2,222	$ 815,824,591
160,000	$ 9,604	$ 2,675	$ 1,964,694,600	$ 8,804	$ 2,675	$ 1,836,635,643	$ 3,906	$ 2,222	$ 980,486,237
180,000	$ 10,308	$ 2,675	$ 2,336,875,637	$ 9,449	$ 2,675	$ 2,182,259,793	$ 4,192	$ 2,222	$ 1,154,528,663
200,000	$ 10,980	$ 2,675	$ 2,731,059,195	$ 10,065	$ 2,675	$ 2,548,052,920	$ 4,465	$ 2,222	$ 1,337,518,780

Table A.58: Global warming and resource depletion indirect impacts from liquid fuels production (functional unit = 1 kg crude oil)

Impact category	Unit	Diesel, at refinery/CH U_SA_Crude_input	Heavy fuel oil, at refinery/CH U_SA_crude_input
Abiotic depletion	kg Sb eq	0.023	0.023
Global warming (GWP100)	kg CO2 eq	0.523	0.487

Table 59: Global warming impact assessment including off-site / indirect impacts

	REVERB.	**FLASH**	**HL-SX-EW**
Direct CO2 emissions	**432,262.06**	**246,717.01**	**275,025.80**
Electricity consumed (kWh)	***851,093,914.85***	***745,457,276.40***	***1,440,176,987.95***
CO2 release from electricity production (tonnes)	859,604.85	752,911.85	1,454,578.76
Diesel consumed (tonnes)	***45,358.94***	***44,876.19***	***75,556.54***
CO2 release from diesel production (tonnes)	23,730.42	23,477.86	39,528.89
Fuel oil consumed (tonnes)	65,565.00	21,824.00	0.00
CO2 release from fuel oil production (tonnes)	31,995.06	10,649.89	0.00

Table 60: Resource depletion impact assessment including off-site / indirect impacts

	REVERB.	**FLASH**	**HL-SX-EW**
Cu ore mining (tonne Sb-eq.)	7.75E+03	6.40E+03	9.51E+03
Diesel production (tonne Sb-eq.)	1065.778665	1054.435735	1775.318121
Fuel oil production (tonne Sb-eq.)	1524.903152	507.5800564	0
Resource depletion (tonne Sb-eq.)	**1.03E+04**	**7.96E+03**	**1.13E+04**

D.2 Case Study 2

Table A.61: Overall gold tailings water balance as a function of feed solids concentration (Option 5 and Option 6 sensitivity analysis)

	Feed solids concentration (wt %)	Feed Water (t/day)	Water Recovered (Dewatering) (t/day)	Water to Tailings from Feed (t/day)	Water to Tailings from Rain (t/day)	Water Loss from Evaporation (t/day)	Water Loss from Seepage (t/day)	Water Recovered (Decanting) (t/day)	Daily Water Accumulation (t/day)	Dissipative Water Loss per year	Dissipative Water Loss per year/10^6
OPTION 6	50	9600	6715	2885	0	3275.4	144.27	1156.00	-1690.17	9.028.E+06	9.03
	45	9600	7034	2566	0	3275.4	128.29	1156.40	-1994.22	1.123.E+06	1.12
	40	9600	6650	2950	0	3275.4	147.51	1156.40	-1629.07	1.130.E+06	1.13
OPTION 5	50	9600	5539	4061	0	3571.7	203.04	1156.00	-869.97	9.965.E+06	9.97
	45	9600	6139	3461	0	3571.7	173.03	1156.40	-1440.56	1.236.E+06	1.24
	40	9600	4124	5476	0	3571.7	273.81	1156.40	474.30	1.269.E+06	1.27

Table A.62: Economic performance of tailings dewatering alternatives as a function of feed solids concentration (Option 5 and Option 6 sensitivity analysis)

	Feed solids concentration (wt %)	Initial Investment	Cash flow	NPV at 13 % discount rate	NPV after 8yrs*
OPTION 6	50	-9,071,755	2,379,776	-6,691,980	-6,691,980
	45	-8,694,982	3,563,063	-5,131,919	-5,131,919
	40	-8,202,729	2,136,735	-6,065,994	-6,065,994
OPTION 5	50	-9,521,677	-66,009	-9,587,686	-9,587,686
	45	-9,752,636	1,890,155	-7,862,481	-7,862,481
	40	-9,752,636	-2,196,022	-11,948,658	-11,948,658

APPENDIX E

Process Models and Detailed Calculations

Please refer to the CD-ROM attached to this thesis for an the detailed process models used for the copper beneficiation case study (Appendix E.1) and the gold tailings dewatering case study (Appendix E.2).